HUMAN IMPACTS ON LIFE IN FRESH WATERS

19

Symposia Biologica Hungarica

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Akadémiai Kiadó, Budapest

HUMAN IMPACTS ON LIFE IN FRESH WATERS

Edited by

P. BÍRÓ and J. SALÁNKI

(Symposia Biologica Hungarica 19)

The volume includes the lectures presented at the Symposium on Human Impacts on Life in Fresh Waters held at the Biological Research Institute of the Hungarian Academy of Sciences, Tihany, between 7–9 September 1977.

The 19 papers by experts from ten countries cover a wide range of limnological, toxicological, physiological and hydrological topics. The volume deals with the cultural eutrophication of fresh waters, the relationships between aquatic plants and animals, and the changes in ecosystems due to human impacts. A major part of the volume has been devoted to the biological effects of pollution of different origin on fresh waters, to their intensity and type and also to the prediction and prevention of their deleterious consequences.

Effects of pollution and the impacts of commercial fisheries on fish fauna or a given fish population both in rivers and stagnant waters are treated in detail. Some effects of toxic compounds on the physiological functions of fresh water invertebrates are also described.

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Edited by

J. SALÁNKI

and

P. BIRÓ

Biological Research Institute
of the Hungarian Academy of Sciences, Tihany, Hungary

AKADÉMIAI KIADÓ, BUDAPEST 1979

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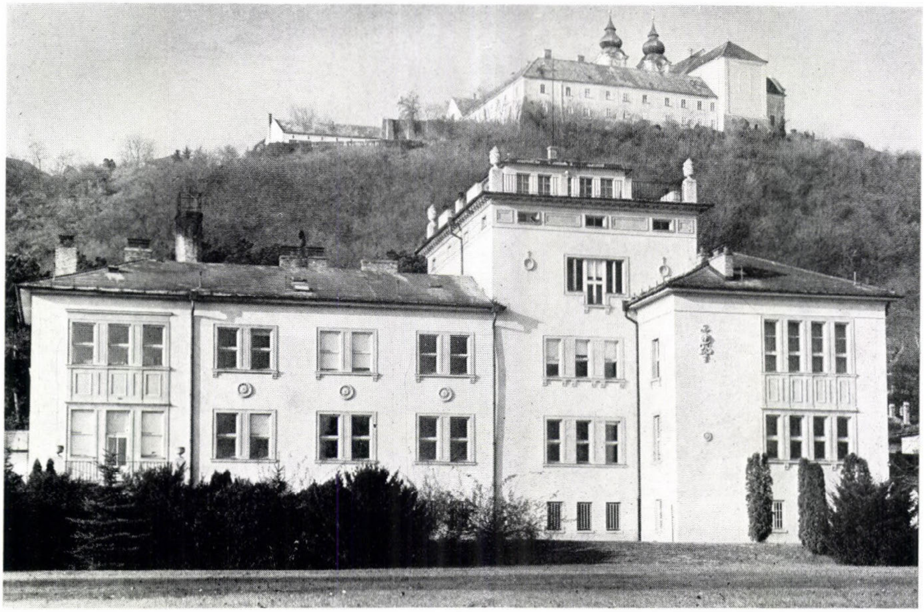

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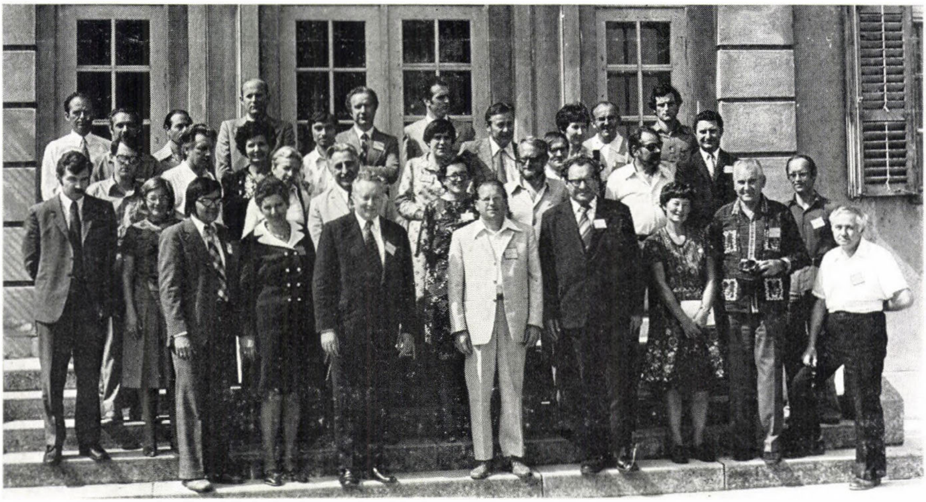

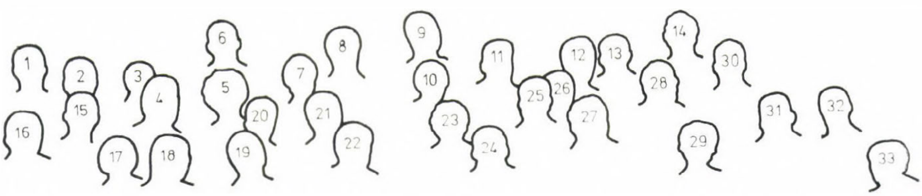

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PREFACE

Among recent problems of environmental biology, human impacts on the life of fresh waters deserve special attention. The papers of the present volume discuss this subject from different aspects, each of them being concerned with a special geographical area, some specific environmental impact and its biological consequences for a natural fresh water ecosystem.

At present, there is an increasing need for clean waters all over the world in face of the growing tendency of pollution that is currently witnessed. The antagonism of these two tendencies must be solved in a way which results in the preservation of the present status of clean lakes and rivers and in the restoration of those having already been polluted.

Biologists may play an important part in planning the actions necessary for the protection of natural waters by examining and exploring the undesirable changes affecting the physiology and composition of the flora and fauna of fresh waters. We hope this volume will be a useful contribution to these efforts.

We would like to express our special gratitude to the contributors. Many thanks also are due to colleagues for their help in organizing the Symposium and for their technical assistance in preparing this volume, especially to Drs B. Entz, S. Herodek, J. Ponyi, Nóra P.-Zánkai, Miss Borbála Horváth and Mrs Judith Komáromi.

The Editors

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OPENING ADDRESS

J. SALÁNKI

DIRECTOR OF THE BIOLOGICAL RESEARCH INSTITUTE OF THE HUNGARIAN ACADEMY OF SCIENCES, TIHANY, HUNGARY

Ladies and Gentlemen,

It is a great pleasure to welcome you to this Symposium held to celebrate the Jubilee of the Biological Research Institute of the Hungarian Academy of Sciences.

When the Biological Research Institute opened its doors on the 5th of September, 1927, it was the only research institute for biology in Hungary. There is, thus, hardly any area within the scope of biology that we have not dealt with here during the past 50 years: botany, genetics, hydrobiology, microbiology, neurobiology, pharmacology, physiology and zoology have all had their turn as the focus of the research work going on.

Today we have two departments, their research concentrating on two main fields: the neurobiology of invertebrates and hydrobiology.

It is, thus, only natural that we should have chosen environmental biology as the topic of this jubilee symposium: both our departments are keenly interested in the effects modern man is having on fresh water life.

But there is yet another reason. Situated as we are on the shore of Lake Balaton, one of the largest shallow lakes of Central Europe, daily we meet here at the Institute a great many problems of water pollution and deterioration of life in our waters.

In view of the concern about the future of our water resources and the environment evident throughout the world, I think we can safely say that we shall be dealing with one of the most urgent problems facing natural sciences today.

I trust that we shall have a chance to hear some of the latest findings on a variety of water ecosystems. May our discussions prove fruitful in ideas for ways of preserving our limited water resources for the use and delight of future generations.

I wish you a pleasant stay in this lovely part of Hungary, and herewith open the Symposium.

Symp. Biol. Hung. 19, pp. 15–20 (1979)

PRESENT AND FUTURE OF LAKE BALATON INTRODUCTORY REMARKS

I. LÁNG

DEPUTY SECRETARY GENERAL OF THE HUNGARIAN ACADEMY OF SCIENCES, BUDAPEST, HUNGARY

In modern times both surface and subsurface waters are becoming increasingly affected by human activities leading to the development of a number of adverse phenomena. Aquatic ecosystems are in the process of undergoing fundamental changes, the quality of water deteriorating in many locations, pollution becoming heavier and resulting occasionally in ecological emergencies.

Our exchange of ideas here, on the shore of Lake Balaton, is expected to prove beneficial to all of us in our future research work. I am pleased to be able to expound on the present and future of Lake Balaton. We would highly appreciate your valuable advice and comments on possibilities of developing our water control system and improving the environment around the lake.

Some data on Lake Balaton, the largest lake of Hungary and one of the most important natural assets of the country, may be of interest (Fig. 1).

The lake is 77 km long, 7.5 to 8.0 km wide, the average and maximum

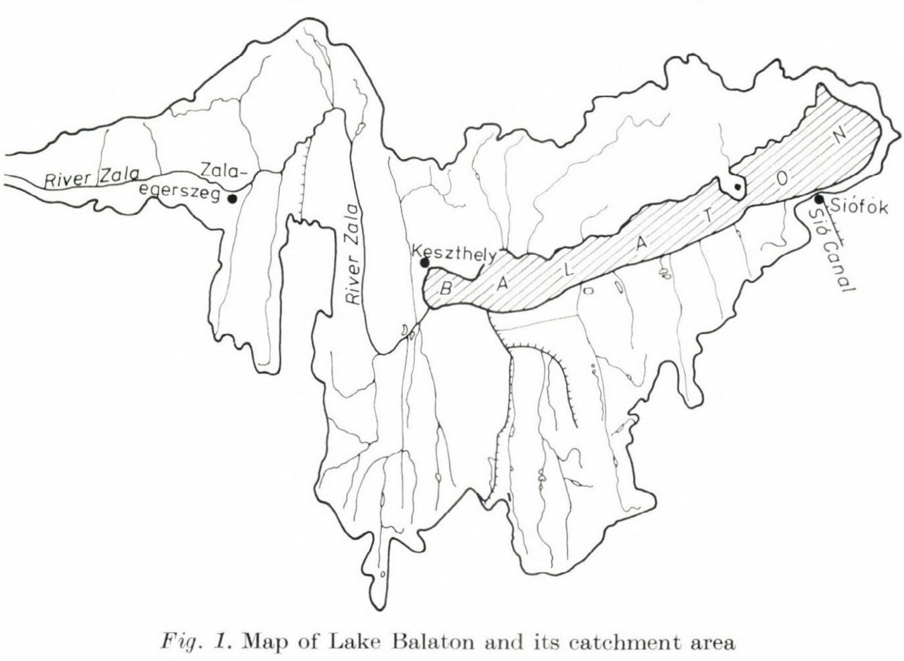

Fig. 1. Map of Lake Balaton and its catchment area

depths are 3 and 11 m, respectively. The surface area is approximately 600 km^2. The catchment area of the lake is rather large, namely 5180 km^2 not counting the lake surface proper. The northern and south-western parts of the catchment area are rich in contours and here the relatively large annual precipitation, i.e. 650 to 700 mm, tends to sustain erosion processes. The catchment is drained by 125 streams to Lake Balaton, of which the largest, the Zala River, contributes about half of the total discharge. The excess water from the lake is released through controlled gates to the Sió Canal and thence to the Danube. The lake and the tributary catchment form an organic water management unit. The aquatic ecosystem of the lake has for centuries been controlled dynamically by the amount and quality of substances entering the tributary streams.

The water of Lake Balaton is eminently suitable for bathing, water sports and recreation. The water has a pH of 8.3, the shallow body of water is rapidly heated, the mean daily air temperature being above 18 °C from mid-June up to mid-September.

Lake Balaton has for a long time been the focus of interest of Hungarian scientists. Around the turn of the century two geographers, Lajos Lóczy and Jenő Cholnoky, wrote and edited, together with several invited experts, a series consisting of 32 volumes and close to 7000 pages entitled "Results of Scientific Studies on Lake Balaton". At that time the work counted among the rare scientific ventures treating the lake and its surroundings as a single unit.

In the late 1920's of the present century, the efforts of Géza Entz and Olga Sebestyén have greatly contributed to exploring the biological conditions of the lake. Experiences accumulating over more than 50 years were summarized in 1973 by Olga Sebestyén in the publication "Problems of Public Interest Related to Lake Balaton", suggesting also various measures considered necessary to protect the lake. For the past 50 years the scientific staff of the Biological Research Institute have systematically been studying the life in and around the lake as well as the calculable impacts of human activities on it. Recent advances in research are going to be reported at the present symposium.

At the Research Centre for Water Resources Development supervised by the National Water Authority, research on the hydrological, hydraulic and, in part, biological characteristics of the lake has been conducted for several decades. Since 1957, the quality of the lake water has been regularly supervised by the regional agencies of the National Water Authority. Samples have been taken at regular intervals from about 60 points including the streams discharging into the lake. Sanitary checks on the water are the responsibility of the county stations of the Ministry of Public Health.

During the past 15 years major changes have occurred in the catchment area of Lake Balaton. Agricultural production has increased greatly as revealed by the following figures: the average yield of autumn wheat in Somogy County between 1961 and 1965 was 1.87 tons per hectare and 3.17 tons per hectare ten years later. The corresponding figures for the counties of Veszprém and Zala are 1.82 and 2.97, and 1.65 and 2.91, respectively. The average increase in yield is 70 per cent. The grain yield of maize increased from 2.5–2.7 to 3.5–3.9 tons per hectare, corresponding to an increase of about 44 per cent. The yield of alfalfa in 1975 surpassed that in

1965 by 23 per cent. These advances have been the results of several beneficial factors. The general standard of farming has improved and fertilizers are being applied at materially higher rates. The total amount of fertilizers applied increased 5.9-fold in Somogy County between 1965 and 1975, and 4.3-fold in Zala County. This increase is, however, significant not only in terms of relative figures but also in those of absolute ones and the application of fertilizers has thus attained a level which may already be termed intensive. The amount of fertilizer substance spread by the state farms over one hectare of land, orchard and vineyard in 1975 was 467 kg in Somogy County, 355 kg in Veszprém County and 429 kg in Zala County. The corresponding figures for the farming cooperatives were 349, 314 and 319 kg, respectively.

Chemical pest control has also advanced tremendously during the past ten years. Herbicides are now used almost exclusively for weed control in maize and cereals. Vines and orchards are sprayed 7 to 8 times during the growing season. The range of pesticides underwent radical changes between 1965 and 1975, biologically more effective products with higher concentrations of active substance having appeared on the market. At the same time, the use of several pesticides has been discontinued. Thus, for instance, pesticides containing DDT as an active agent have not been used since 1970 in the catchment area of the lake.

Livestock breeding has started to concentrate on specialized animal farms. Presently, there are 19 such farms in the catchment area producing large volumes of liquid manure. At these farms approximately 40 thousand pigs and 3 thousand head of cattle are kept. No liquid manure is allowed to be discharged into open recipients and specialized farms are continuously faced with the difficult task of safe disposal. Licenses for the construction of new farms are not granted unless the disposal of the liquid manure is technically guaranteed.

Another important change is associated with the rapid increase of tourism. Lake Balaton has become very popular with the Hungarian public. Many visitors from abroad, especially from Czechoslovakia, Austria, the German Democratic Republic, the German Federal Republic and from the Scandinavian countries seek summer recreation along the shores of the lake. In the three counties of Somogy, Veszprém and Zala surrounding the lake (Fig. 2), the total number of guest-days spent at the commercial tourist accommodations was 700 thousand in July 1965 and over 2 million in July 1975, representing an almost threefold increase. During week-ends in the summer months, about 600 thousand people visit the area. Fifteen years ago this number was half of this figure. Domestic water supply has kept pace with the growth of tourism, while sewerage and sewage treatment have failed to do so.

Among the industries a few food processing plants give only cause for concern. Their effluent treatment plants are imperfect and considerable amounts of phosphorus and nitrogen are still discharged from these plants into the lake.

As a result of the above changes a new situation has emerged in the catchment area of Lake Balaton over the last decade. The first alarm was raised in spring, 1965, triggered by a mass decay of fish. The accumulation of pesticides belonging to the chlorinated hydrocarbon type has been identified as the most probable cause of this disaster, greatly contributing to the ban on DDT in Hungary at the end of the 1960's. A local fishkill occurred in spring, 1975, which could, however, be attributed to ecological conditions, specifically to the depletion of oxygen in the water. Nevertheless, the Hungarian public expressed great concern over this second fishkill, since environmental protection has in the meantime become a matter of public interest.

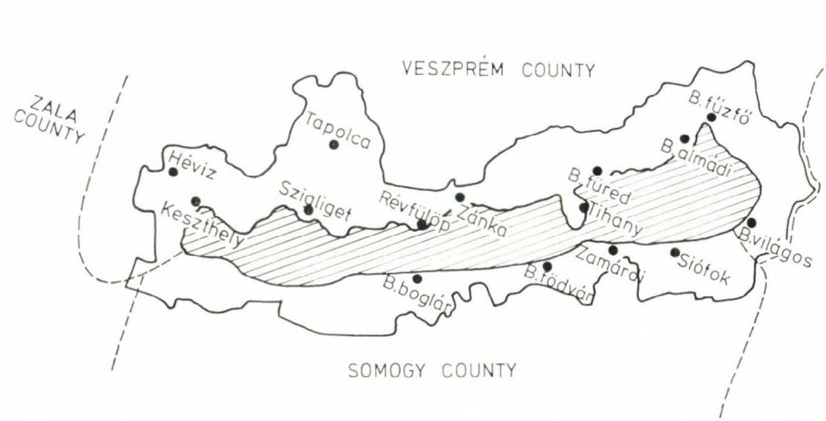

Fig. 2. Recreation zone of Lake Balaton

A brief review of the organization of environmental protection in Hungary is considered necessary at this point in order to give a better insight into the purposes and scope of these measures.

The first institutional measures for environmental protection date back to the period between the two world wars in Hungary. The legal foundations for pollution control and land conservation were laid in the early 1950's. Environmental protection, as a popular movement, a complex scientific project, as well as the basis of formulating and realizing regional development plans started in this country in 1970. Lengthy debates were conducted on whether or not to organize a ministry for environmental protection. Eventually it was agreed on to make the respective ministries responsible for enforcing environmental measures in their particular spheres of competence.

The National Council for Protection of the Environment, as a coordinating, consulting and supervisory agency of the Council of Ministers was founded in 1974. The Council is composed of 30 members, including representatives of all ministries and social organizations concerned, thus also of the Hungarian Academy of Sciences. The resolutions passed by the National Council for Protection of the Environment are approved and enforced by the Council of Ministers. A bill on the protection of the human environment was passed by the Hungarian Parliament in 1976. This Act contains comprehensive provisions on the scope of environmental protection in Hungary, as well as on responsibilities, and has introduced the concept of criminal prosecution for environmental offences.

In Hungary the principal research objectives up to 1990 have been identified in the National Perspective Scientific Research Plan. Environment-oriented research figures among the 17 programmes enjoying outstanding national research priority.

A number of agencies are engaged in the environmental problems related to Lake Balaton. Of these, the Executive Council for Lake Balaton should be mentioned first as an independent agency with certain funds at its disposal. Important recommendations and proposals are submitted by this body to the Government and to the local authorities. The Council has set up a sub-committee on environmental protection, the activities of which are greatly appreciated. The regional branch of the Hungarian Academy of Sciences at Veszprém also deals regularly with the environmental problems of Lake Balaton. Items concerning the quality of water in the lake have repeatedly appeared on the agenda of various sessions of the Hungarian Hydrological Society.

After the mass fish decay in 1975, the National Council for Protection of the Environment passed a resolution on intensified coordination and extension of research work related to Lake Balaton. The Commission for Coordinating Environmental Research on Lake Balaton was formed, comprising the representatives of all agencies concerned. The Coordinating Commission is supervised by the Hungarian Academy of Sciences, where I have been assigned to the honourable task of heading the Coordinating Commission. The organizational and administrative tasks essential to the functioning of the Coordinating Commission are taken care of by the Biological Research Institute at Tihany. The Coordinating Commission has reviewed the state of all research projects in Hungary which are in any way related to environmental protection in the Balaton Region. Three spheres of research were assigned priority by the Coordinating Commission in 1975, namely (*i*) Studies into the parameters describing, and the factors deteriorating or improving, the quality of the lake water; (*ii*) Research in the domains of economics, law and other social sciences to prepare environmental decisions in the region; and (*iii*) Studies on the environmental factors affecting recreation and tourism in the recreational areas.

A resolution has been passed by the Coordinating Commission to realize two additional important projects, namely (*i*) The review and synthesis of the major scientific results and professional experiences obtained so far concerning the region, and (*ii*) Development of the scientific foundations for an integrated environmental monitoring system.

The earlier scientific data concerning the environment of Lake Balaton have been processed. Based on these, the Coordinating Commission has formulated twenty recommendations primarily as regards environmental protection. These long-term plans call for top-level decisions and major investments. Work on the development of the scientific principles of the integrated monitoring system has been started recently and the first results are expected in about a year's time.

The fundamental environmental problem of Lake Balaton is the acceleration of eutrophication phenomena. Primary biological production has intensified in the lake. This problem will form the subject of a special report at the Symposium. The amount of nitrogen and phosphorus discharged into the lake has increased substantially as compared with earlier periods. The sources of this nutrient supply are the fertilizers used in agriculture, the soil particles carried into the lake by erosion processes, moreover the effluents and refuse from settlements. Some food processing plants also pass important amounts of nitrogen and phosphorus to the lake water. These nutrients are essential elements in supporting life. They are not toxic, like some pesticides or heavy metals, still may cause heavy economic losses by changing the composition of the original aquatic ecosystems and by impairing the quality of water. The primary objective of future environmental protection in the Balaton region is to minimize the amount of such substances, specifically of nitrogen and phosphorus, entering the lake. There are evidently other environmental objectives as well, e.g. reduction of the noise level in recreational areas, the abatement of air pollution caused by transport, nature conservation, etc.

The fundamental function of Lake Balaton is to meet the demands for recreation. Intensive farming in the catchment area is similarly essential, since agriculture contributes an important share to the national income of Hungary. Restriction of farming is conceivable in exceptional cases only, there being no reserve lands available in the country. These demands are conflicting to a certain extent. Compromises are evidently needed and the occasionally competing interests of the individual sectors, such as tourism, agricultural production, etc. should be reconciled.

The water in Lake Balaton is still eminently suitable for bathing, except for the Bay of Keszthely at the southwestern end, where its quality has already deteriorated perceptibly.

Environmental protection around Lake Balaton forms part of the regional development plans. To ensure systematical, controlled development in the surroundings of the lake, a regional plan for the development of Lake Balaton was accepted in 1957, followed by the Central Development Programme for Lake Balaton in 1969, in which the major development objectives were established for five-year cycles. The second phase (1976–1980) of this programme is presently under way, the policy and objectives of the third phase (1981–1985) being prepared for approval by the Council of Ministers.

In formulating the perspective policy, the recommendations compiled by the Commission for Coordinating Environmental Research on the basis of existing scientific evidence have provided valuable guidance.

The development of water management represents a special task within the central development programme. The plan is highly ramified in that it comprises regulation of the lake bed, control of the shorelines, water surface regulation, and maintenance of water quality. Great importance should be attributed to the perspective policy of providing facilities for sewage treatment to all settlements and to develop the sewerage system in a way permitting as far as possible all effluents to be removed from the catchment area. The building of an artificial reservoir is contemplated at the mouth of the River Zala in which the major part of the soil eroded from the catchment area and of the plant nutrients, will be retained. Realizing the ambitious regional development objectives, heavy investments over three five-year plan periods are expected to be necessary. If we succeed in realizing these plans by 1995, environmental protection in the Lake Balaton area will also be ensured for the more distant future.

Scientific research plays an important role in laying the foundations for these development plans, in selecting the optimal alternatives, and in predicting the long-range effects.

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PESTICIDE RESIDUES IN LAKE BALATON

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Abstract

No residues of chlorinated hydrocarbons, e.g. DDT and metabolites and BHC isomers, have lately been found in samples of water, plankton, mussel and fish (bream, *Abramis brama* L. and carp, *Cyprinus carpio* L.) from Lake Balaton. In the liver and fat of pikeperch (*Stizostedion lucioperca* L.) lindane has been found (10–30 ppb) but this value is not statistically significant because of the limited number of samples. The lake seems to be free of chlorinated insecticide residues due to the total ban of DDT and other chlorinated type insecticides in 1970. A residue of 2,4-D has been found in the water samples collected from the lake.

INTRODUCTION

The first serious fishkill in Lake Balaton occurred in 1965 (Baron et al. 1967). Cieleszky and Dénes (1965, unpublished data; see Baron et al. 1967) determined 0.1–8.4 ppm DDT in predatory fishes and 0.1–0.8 ppm DDT in nonpredatory fishes. After this fishkill extensive analytical research work has been started to determine the various pesticide residues in water and aquatic organisms, with special attention to those of the chlorinated hydrocarbon type.

The first detailed study was published by Baron et al. in 1967. These authors investigated the residues of DDT and DDT decomposition products, isomers of BHC, etc., applying thin-layer chromatographic technique. Table 1 shows the most characteristic results of this investigation. They

TABLE 1

Residues in Fish and Aquatic Organisms (ppm)
(Data of Baron et al. 1967)

	DDT	BHC
Crustacean plankton	0.1 –0.2	0.03–0.05
Mussel	0.15–0.25	0.1 –0.15
Carp		
meat	0.08	0.02
liver	0.05	0.02
fat	0.1 –0.3	0.02
Pikeperch		
meat	0.1 –0.3	0.05
liver	0.2 –0.3	0.1 –0.2
fat	0.4 –1.0	0.3 –0.7

did not find Aldrin and Dieldrin residues. It was interesting that in those years the same amounts of DDT and BHC residues were found in fishes taken from the River Danube.

In the food web of Lake Balaton the zooplankton has a significant role. In 1967, Ponyi et al. (1968) investigated the chlorinated hydrocarbon content of crustacean plankton in five sections of Lake Balaton. They found 0.2–2 ppm residues of DDT and γ-BHC.

TABLE 2

Results of Water Analyses

Pesticide	Year	Residue (ppb)	Reference
Lindane	1967	0.01–0.08	Pinkola and Tóth (1971)
	1968	0.09–0.18	Pinkola and Tóth (1971)
	1969	0.08–0.15	Pinkola and Tóth (1971)
	1971	0.01–0.1	Czeglédy-Jankó et al. (1973)
2,4-D	1971	0.01–0.1	Czeglédy-Jankó et al. (1973)
Chlorotriazine	1976	0.01–0.02	Károly and Füzesi (1977)

The chlorinated hydrocarbon content of the water was investigated by Pinkola and Tóth (1971) between 1967 and 1969 (Table 2). They found 0.1–0.2 ppb lindane residue on an average by gas chromatography with EC detector extracting 5 l of water with hexane. Other results of water analyses are shown in Table 2. Károly and Füzesi (1977) investigated 149 samples of water in the spring of 1976 by gas chromatography and thin-layer chromatography for residues of chlorinated hydrocarbons, organophosphorus compounds, dinitrophenol derivatives, chlorophenoxy-acetic acids and chlorotriazines. They could detect chlorotriazine residue altogether in 10 samples, on a level of 10–20 ppb.

TABLE 3

Detection Limits of Pesticides (μg)

Pesticides	PC	TCL	GC
Chlorinated hydrocarbons	0.5	0.01	0.001
Organophosphate compounds	1	0.2	0.01
Phenoxy-acetic acids	0.5		0.05

PC = paper chromatography; TLC = thin-layer chromatography; GC = gas chromatography

In Table 3, the detection limits of various pesticide-groups are given in μg. The values were obtained by gas chromatographic techniques with selective detectors, taking an extraction of 5 l of water evaporated to 1 ml, injecting 1 μl into the gas chromatograph. The results obtained indicate that analytical data on the pollution of the water of Lake Balaton published in the literature refer to results determined on the level of detection limits, or in some cases below them. Therefore the criticism of these data is justified.

Consequently, we deal with the question of identification of the individual compounds, one of the greatest problems of pesticide residue analysis at present.

MATERIAL AND METHODS, COLLECTING SITES

In 1976–1977 the residues of chlorinated hydrocarbon, DDT and decomposition products, isomers of BHC, organophosphates and phenoxy-acetic acid were investigated in water, plankton, mussel, bream (*Abramis brama* L.) and pikeperch (*Stizostedion lucioperca* L.).

The chlorinated hydrocarbon residues were determined by the gas chromatographic technique using an EC detector after homogenation with anhydrous Na_2SO_4, extraction with n-hexane, evaporation and acetonitril partitioning. For the determination of chlorinated hydrocarbons 5 l water was extracted with n-hexane. The water–solvent ratio was 5 : 1. After evaporation the sample was injected directly into the gas chromatograph and analysed.

For determination of organophosphate residues the water sample was directly investigated by the acetylcholinesterase enzyme inhibition technique, after extraction with n-hexane and evaporation the AChE method was and again applied, both directly and after oxidation with bromine.

The residue of 2,4-dichlorophenoxy-acetic acid was determined by its ultraviolet absorption spectra after extraction and clean-up. Figure 1 shows the most important steps of 2,4-D residue analysis. The most sensitive analytical method known today was used.

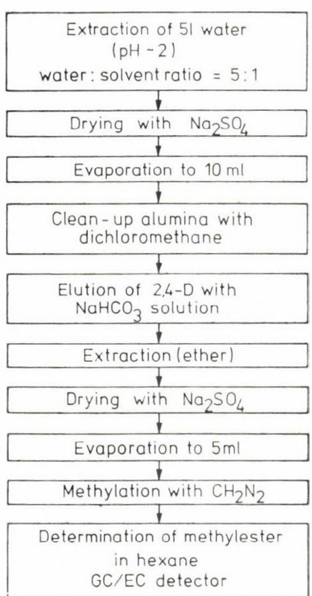

Fig. 1. Determination of 2,4-D in water

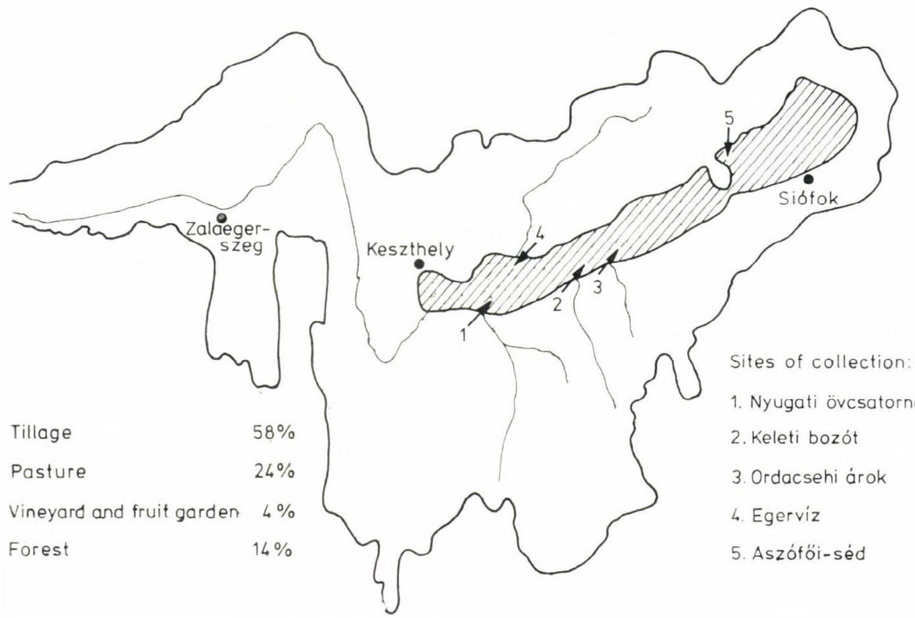

Fig. 2. Catchment area of Lake Balaton

The sites of water sample collection are given in Fig 2, which shows that 84 per cent of the catchment area of Lake Balaton is under cultivation with a significant use of herbicides.

RESULTS AND CONCLUSIONS

In aquatic organisms no residues of organophosphates and 2,4-D were found. However, γ-BHC (lindane) was determined: in mussel 10 ppb, in bream liver 8 ppb, in pikeperch liver 20 ppb and in fat 30 ppb. To confirm the identity of γ-BHC a double check after hydrolysis of extracts was made, and the peak of γ-BHC did not appear. But these results are not statistically significant because of the limited number of samples. However, our results show the drastic decrease in the level of pesticide contamination as compared with data before 1970, owing to restrictions and a ban of the use of chlorinated hydrocarbon-type insecticides.

In 1976 water samples were investigated from 2 different locations of Lake Balaton. Gas chromatogram produced peaks with the same retention times as shown by BHC isomers, DDE and DDT. However, when attempts were made to confirm the identity of these compounds by the gas chromatograph-mass spectrometer system (Varian MAT 111, Department of Organic Chemistry, Technical University of Veszprém, Hungary), it appeared that these peaks did not correspond to any chlorinated hydrocarbon, meaning that there was no chlorinated hydrocarbon insecticide residue in the water of Lake Balaton. In our experiments the detection level in concentration was 0.01 ppb for lindane and 0.02 ppb for DDT.

No residues of organophosphate compoate were found either. Detection level was 1 ppb.

In samples taken in September and November 1976, 2,4-D was found: about 40 ppb in September, and about 25 ppb in November. Ultraviolet absorption spectra and also colorimetric analysis with chromotropic acid were applied in these tests. In the period between February and June 1976 however, Károly and Füzesi (1977) did not find any residue. Consequently investigations were continued for determining 2,4-D residue every month in 1977.

Results of 2,4-D residue determinations are shown in Table 4. Sample numbers refer to the appropriate sites of sample collection represented in Fig. 2. Samples were collected from each place at the mouth of the respective creeks (*a*), 100 m from the mouth (*b*) and 500 m from the mouth (*c*).

TABLE 4

Results of 2,4-D Analyses in the Water of Lake Balaton (ppb)

Sample*		23rd February 1977	22nd March 1977	12th April 1977	6th May 1977	1st June 1977	1st July 1977
1	*a*			2	1		
	b					1	2
	c			4			4
2	*a*			2		3	5
	b			2	2	4	1
	c			1			1
3	*a*				2		2
	b					2	1
	c						1
4	*a*				4	4	5
	b				2	4	1
	c				4	6	1
5	*a*				2	5	
	b				2		10
	c				4		2

* For sites *see* Fig. 2 and text.

In the first quarter of the year 2,4-D residue could not be detected. In agricultural practice, the usage of 2,4-D begins in April and May under normal meteorological conditions but the usage of herbicide combinations containing 2,4-D begins, in some cases, at the end of March. Table 4 demonstrates that the 2,4-D residue could be detected in the water in five samples in April, in 9 samples in May but in 13 samples in July. It is probable that one or two months are necessary for 2,4-D to enter the water following a rainfall.

SUMMARY

On the basis of investigations made in 1976 it seems there are no residues of chlorinated hydrocarbon and organophosphate-type insecticides in Lake Balaton, as also shown by Károly and Füzesi (1977).

Reports on residues of 2,4-D should be attended to because the detectable level (1–10 ppb) may have a hormonal effect on various plants or organisms.

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SYSTEMATIC CLASSIFICATION AND CHARACTERIZATION OF WATER POLLUTANTS

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Abstract

In the absence of systematic water management both as regards the quality and quantity of natural waters, an increasing influence of pollutants is a real hazard. In view of the heavily strained water resources of the G.D.R. effective legislation as well as adequate information about the most significant water pollutants, particularly pesticides, are urgently needed. This will enable producers as well as users to assess any potential damage that might be caused to any individual body of water. The methods applied in investigating the different pollutants as well as the structure of a catalogue containing the characteristic data of water pollutants are presented.

Water pollutants are prominent among the factors having an adverse effect on natural bodies of water. Water pollutants are inorganic or organic substances which get into the surface or ground-water changing its state in such a way that its capacity for self-purification and the productivity of surface waters is affected, the use of water by the population and industry becomes difficult or impossible, and the recreational value of waters is reduced.

If no systematic qualitative and quantitative management of the water resources is ensured, there is a danger of increasing amounts of pollutants being discharged into the bodies of water as a result of rising living standards, the construction of new industrial plants, a more extensive use of fertilizers and pesticides in agriculture, transition to industrial-type animal husbandry, and the wider use of artificial irrigation. Therefore, it is necessary, in addition to the construction of sewage treatment plants, to use fertilizers and pesticides on the basis of accurate planning. Parameters like the most favourable form of application, toxicity and persistence must be accurately known.

Special attention should also be paid to the way in which irrigation water is used in agriculture (timing, single doses, soil type, crop).

The water resources of the G.D.R. are extremely strained as the country has an area of only 108 000 km^2 but a population density of 155 inhabitants per km^2 and a large industry and agriculture.

Even if considering the total potential water resources (including the inflow from abroad by way of the Elbe), the mean rate of use is 31 per cent already, and the water of the rivers is used several times over; in some rivers the same water is recycled five times. The water management authorities of the G.D.R. pay much attention to the problem of water pollutants on the basis of the existing legislation.

The Ministry of Environmental Protection and Water Management has issued a list of pollutants. From time to time this list is updated by a board of experts. This board, composed of representatives of the authorities of water management, environmental protection, health and industry, has also selected a number of economically significant substances from among the multitude of water pollutants for experimental investigations as regards their adverse effects on waters. It is due to this approach, but also to a close cooperation with the chemical industry, that no pesticide or detergent is released for production and use without having been subjected to toxicological and biochemical tests.

These tests have been carried out in the Berlin Institute of Water Management for some 15 years.

Initial sporadic work has increasingly developed into systematic efforts aimed at creating and publishing a catalogue containing the characteristic data of water pollutants. The result of these efforts is the catalogue of water pollutants which, in its first version, describes about 250 substances significant for the national economy, and is to be completed with data on other substances until 1980. Cooperation with a Soviet institute helped to reduce the amount of work required for this project. The methods applied in the investigations and the structure of the catalogue are described below.

METHODS OF INVESTIGATION USED TO DETERMINE CHARACTERISTIC DATA

The central laboratory of the Institute of Water Management carries out investigations in order to characterize water pollutants as regards

1. Toxicity
2. Biochemical degradability
3. Chemical oxidizability
4. Adsorbability
5. Flocculation

Ad 1. In the toxicological water tests the β- to α-mesosaprobic green alga, *Ankistrodesmus falcatus* (Corda) Ralfs, the scattered carp-fingerling (*Cyprinus carpio* L.) and *Poecilia reticulata* are mainly used. These tests serve to determine the following four criteria of toxicity: (*i*) critical concentration under laboratory conditions; (*ii*) threshold concentration; (*iii*) 50 per cent retardation of propagation or mean lethal concentration (LC_{50}); (*iv*) lethal concentration (LC_{100}).

The algological tests are carried out in accordance with the *Ankistrodesmus* test, and the mean lethal concentration is determined graphically by the probit analysis (Legler 1970 and 1972). The general assessment of the toxicity of a test substance in water is based on its threshold concentration value.

Ad 2. The investigation of water pollutants with regard to their biochemical degradability, which is a good index of the harmful effects caused by a substance in a body of water and of its removal from sewage, is carried out both in continuously operated laboratory models and in the respirometer. For a period of several weeks the microflora of the activated sludge is adapted to the chemical under study, while also adding substances and nutrients. After successful adaptation an extensive programme of analyses is carried out in order to determine the rate of decomposition and its dependence on various parameters.

Adapted activated sludge is also used to analyse the decomposition process in the Warburg respirometer (Legler 1970 and 1972). Biochemical degradability is assessed based on all the test results, including those obtained with non-adapted material.

Ad 3. To determine the chemical oxidizability, chlorine and ozone, the most common oxidizing agents employed in drinking water treatment, are used. The oxidizing agent is added in doses of 50 mg to an aqueous solution of the pollutant to be investigated; the concentration of the oxidizing agent ranges from 50 to 300 mg per l. After a defined time the reaction effect is assessed. The time of reaction is 1 to 24 hours in the case of chlorination; as for ozonization, the measurements are taken immediately after gassing has been stopped. Besides the determination of totals, colorimetric, spectrometric and chromatographic methods are used.

Ad 4. The extent to which organic matter is adsorbed by powdered and granular activated coal as well as by adsorber resins and ion exchangers is determined in discontinuous agitating tests and continuous filtering tests. The adsorption effect is evaluated on the basis of balanced loading which is graphically derived from the Langmuir isothermal curve, the slope tg of the isothermal curve characterizing the kinetics of adsorption, and the purification effect RE 500 (use of 500 mg per l of adsorbent) if powdered coal is used.

Ad 5. For the flocculation tests the common inorganic flocculants—aluminum sulphate, aluminum oxychloride, ferrous chloride—as well as mixtures of chemicals constituting aqueous suspension mixtures of hydroxions and hydroxides are used. A starting bath about four weeks old is used to obtain comparable basic values. The flocculants are added in doses of 200 mg per l each up to a maximum of 1000 mg per l, and the pH value is regulated. The purification effect is determined by measurements of totals or of specific factors following centrifugation or filtration of the treated water.

CATALOGUE OF WATER POLLUTANTS

The Institute of Water Management (central laboratory), acting on behalf of the Ministry of Environmental Protection and Water Management of the G.D.R., has compiled a catalogue of water pollutants (Wotzka 1975) which is the result of intensified research undertaken in the fields of environmental protection and water management. It contains the results obtained in the central laboratory by means of the above-mentioned testing methods as well as characteristic data which were made available by partner institutions or derived from the literature.

At the present level of our knowledge it seemed most expedient to choose well-defined single substances to form the basis of our system since relatively little is known about summational, cumulative, synergistic or antagonistic effects which may occur in mixtures of substances.

A pollutant is characterized by 14 items with corresponding subitems:

1. Chemical symbol or name of the commercial product
2. Classification by groups of substances
3. Structural formula or characterization of the commercial product
4. Important G.D.R. products containing the substance concerned
5. Solubility in water
6. State of aggregation, appearance, organoleptic properties, melting and boiling points
7. Degradation mechanisms and possibilities of removal
7.1 Biochemical degradability
7.2 Chemical oxidizability
7.3 Adsorbability
7.4 Flocculation
8. Toxicity to warm-blooded animals
8.1 Acute toxicity
8.2 Semi-chronic or chronic toxicity
8.3 Cancerogenic effects
9. Hygienic and other limiting or recommended values
10. Toxicity to aquatic organisms
11. Analytical instructions
12. Special remarks
13. General assessment
14. Bibliography

Data on each substance are summarized under item 13 which classifies it, according to its general dangerousness, into one of three categories:

I: very dangerous pollutant
II: dangerous pollutant
III: pollutant presenting little danger.

As the above definition shows, category III does not comprise pollutants in the strict sense, but the compilation of characteristic data of such substances is also very important.

The appearance of the catalogue of water pollutants in the form of a series of files, the first part of which has already been released, permits its constant updating and facilitates its use: (*i*) obtaining toxicological and biochemical limiting values regarding both the possible treatment of characteristic sewage constituents and the pollution load capacity of receiving waters in relation to these compounds; (*ii*) basic technologies can be devised from the characteristic data for the degradation mechanisms and removal of pollutants; (*iii*) recommendation of the manufacture and use of substances which do not affect water quality; (*iv*) water management authorities can establish limiting values based on the data of the catalogue; (*v*) its use in preparing fundamental decisions (planning, forecasting, legislation).

Thus the catalogue of water pollutants is a means to solve scientific and practical problems regarding the improvement of water quality and the protection of various bodies of water by identification and characterization of the pollutants.

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ACCUMULATION AND RELEASE OF NUTRIENTS BY AQUATIC MACROPHYTES

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Abstract

The accumulation of K, Na, Ca, Mg, Mn, Zn, Fe, P and N was studied in a bog plant and in aerohydatophytic and submerged aquatic macrophytes. The lowest amounts of both macro- and micronutrients were found in the bog plant. More nutrient was accumulated by the aerohydatophytes and submerged macrophytes, but the highest amounts were detected in the ones with submerged roots. In these plants the Mn and Fe content frequently differed by two orders of magnitude. These results prove the importance of aquatic vegetation in water purification. However, if the plants are not removed before decomposition, the nutrients return into the environment. Therefore, we have also studied the process of decomposition in terms of nutrient release, and discuss their role in eutrophication.

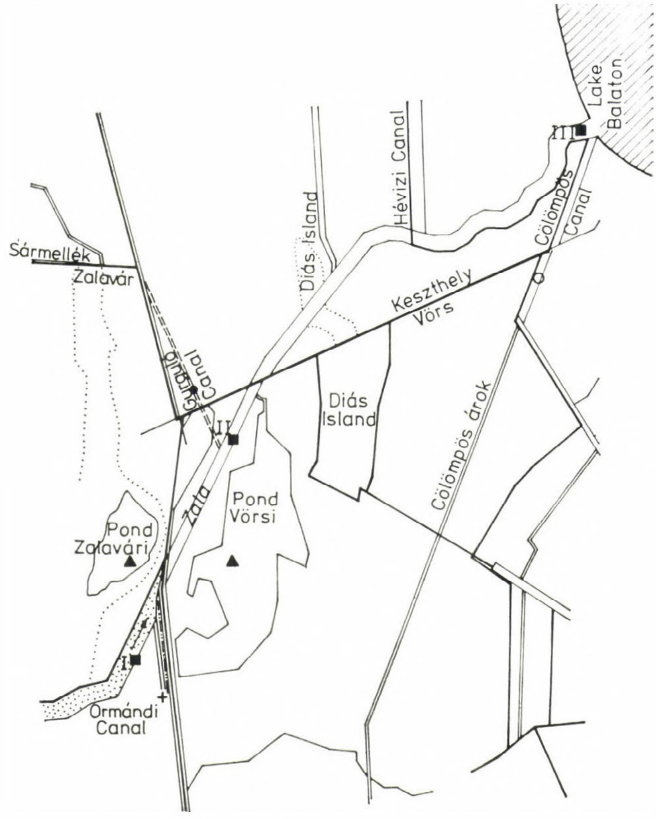
INTRODUCTION

In a previous publication we have dealt with some aspects of the accumulation of micro- and macronutrients in aquatic macrophytes (Kárpáti et al. 1967). The present investigations were carried out in the nature conservation area of Kisbalaton and in the Keszthely and Szigliget bays of Lake Balaton (Fig. 1). Due to intensive agriculture in the water catchment area, the nutrient loads of Kisbalaton, and of the western bays of the lake are very high, resulting in rapid siltation and eutrophication. If before decomposition the water plants are not removed, the accumulated biogenic elements are released into the water, thus increasing its nutrient content. Therefore, it appeared necessary to study the rate of decomposition, too.

Contrary to natural eutrophication, which is a slow ecological transformation, cultural eutrophication resulting from large nutrient loads is a very rapid process requiring some decades. The canal system of Kisbalaton and the River Zala transport large amounts of organic matter and fertilizers into the studied area. This is proved by the propagation of certain macrophytes characteristic of areas affected by man.

METHODS

Stands of macrophytes were sampled by the 'water monolith' method (Kárpáti and Varga 1970). The monoliths were delimited by frames of 1 m^2 area, and all plants were collected from them. After blotting, the fresh weight of plants was determined. The material was dried to constant weight at 105 °C, and the dry weight was measured. These values were used by comparing the accumulation and release of elements to units of area.



Sites of collection

1. Zala ■ I	5. Pond Zalavári ▲
2. II	6. Pond Vörsi ▲
3. III	7. Gurguló •
4. Ormándi Canal +	8. Cölömpös o

Fig. 1. Map of the nature conservation area of Kisbalaton

K, Na and Ca were determined by the Flapho-4 flame photometer. Mg, Zn, Mn and Fe were determined by the AAS-1 atom absorption spectrophotometer. The ash of plants was dissolved in $HCl-HNO_3$. The values are compared to fresh weight. After dissolution, the total P content was determined by the molybdenum blue reaction (di Gléria), using a spectrophotometer. The total N content was measured by the micro-Kjeldahl–Winkler boric acid method. The soil and water samples were analysed by the same methods but in this case, the material was treated with ammonium lactate.

Release of biogenic elements during decomposition of the plants was studied by Rapp's (1967) percolation technique. The percolation tubes contained 20 g plant material and 100 ml distilled water. They were kept in thermostat at 28 °C. The water was changed after 20 minutes, after one week, then monthly. The collected effluents were analysed for the biogenic elements as described above.

Of the characteristic plants of the area, 17 species were investigated according to the following ecological grouping.

Aerohydatophytes

Floating: *Spirodela polyrrhiza*
Lemna gibba
Lemna minor
Wolffia arrhiza
Hydrocharis morsus-ranae
*Stratiotes aloides**

Rooted: *Nymphaea alba*
Nuphar lutea
Trapa natans
*Stratiotes aloides**

Submerged plants

Floating: *Lemna trisulca*
Ceratophyllum demersum
Cladophora sp.

Rooted: *Potamogeton perfoliatus*
Potamogeton pectinatus
Myriophyllum spicatum
Najas marina
Anacharis canadensis
*Stratiotes aloides**

The samples were collected between July 25, 1975, and August 10, 1975, i.e. during the period of the optimal development of water vegetation.

RESULTS

The results of water and mud analysis (Table 1) serve only as an ecological background. The nutrient accumulation in the plants is discussed in detail (Table 2). For comparison the composition of the bog plant *Sagittaria sagittifolia* is also published here.

Data of Table 2 show that the content of the investigated elements, expressed in mg per 100 g fresh weight, is in all cases the lowest in the bog plant. The only exception is the total P content, which is higher than the average. Both the floating and rooted aerohydatophytes accumulate more K than the submerged plants. The Na levels were similar in the investigated plants. The Ca and Mg values of the plant groups did not differ too much. On the other hand, the micronutrients (Mn, Zn, Fe) were found in greatly differing concentrations. The lowest concentrations were found in the bog

* *Stratiotes aloides* accommodates to the water depth and, therefore, belongs to several groups.

TABLE 1

Results of Mud and Water Analysis

May	Total-N		Total-P		K		Na	
	soil mg/100 g	water mg/l	soil mg/100 g	water mg/l	soil mg/100 g	water mg/l	soil mg/100 g	water mg/l
Zala I	1593	2.67	470	0.237	8.9	6.82	14.0	28.2
Zala II	3378	3.53	1040	0.320	11.1	6.25	12.3	26.2
Zala III	2199	3.33	1070	0.257	—	7.12	—	26.8
Ormándi Canal	1448	4.20	344	2.000	7.1	2.55	17.0	12.8
Pond Zalavári	965	5.10	132	0.875	9.6	4.64	75.0	8.8
Pond Vörsi	3376	7.20	374	0.550	12.5	4.90	17.0	22.4
Gurguló Canal	1930	3.80	400	1.160	8.7	5.40	27.5	14.2
Cölömpös Canal	2896	3.20	253	2.000	3.3	6.50	23.5	16.0
October								
Zala I	3378	2.43	526	0.273	18.5	10.25	7.3	22.0
Zala II	4826	2.86	1120	0.253	19.4	9.05	30.3	18.6
Zala III	4129	3.77	1150	0.310	15.7	9.38	14.8	18.7
Ormándi Canal	9169	2.87	416	0.625	14.5	4.40	4.5	8.6
Pond Zalavári	2413	6.80	303	1.120	10.9	1.10	81.0	7.9
Pond Vörsi	3204	3.46	218	0.250	17.2	3.80	86.5	16.4
Gurguló Canal	3861	3.60	498	1.425	16.2	6.30	11.8	13.6
Cölömpös Canal	—	1.30	—	0.230	—	6.35	—	20.1

TABLE 2

Results of

Species	P dry weight	Ash content	K	Na
	%		mg/100 g	
BOG PLANT				
*Sagittaria sagittifolia***	15.02	1.85	48.95	2.15
AEROHYDATOPHYTES				
Floating:				
Spirodela polyrrhiza+	8.42	1.21	13.75	5.13
Lemna minor + *L. gibba*+	7.03	1.63	16.33	3.95
Lemna minor+	4.63	0.69	52.63	1.10
Hydrocharis morsus-ranae+	9.23	1.34	37.25	13.90
Stratiotes aloides+	11.01	2.05	51.88	7.63
Rooted:				
Nymphaea alba++	12.39	1.19	17.25	20.50
Nuphar lutea++	12.11	1.34	46.63	3.63
Trapa natans++	14.64	1.25	73.13	5.25
SUBMERGED MACROPHYTES				
Floating:				
Lemna trisulca++	9.03	1.93	12.88	2.73
*Ceratophyllum demersum***	13.01	2.05	26.50	6.88
Cladophora sp.++	8.75	1.89	27.00	1.50

+Pond Zalavári ++ Pond Vörsi; * Zala II (canal draining. Pond Vörsi); ** Zala III

(Kisbalaton 1975)

Ca		Mg		Mn		Zn		Fe	
soil mg/100 g	water mg/l	soil mg/100 g	water mg/l	soil mg/100 g	water mg/l	soil mg/100 g	water mg/l	soil mg/100 g	water mg/l
1025	82.5	191	29.5	25	—	1.2	below	82.2	below
1238	85.5	214	27.7	29.5	—	2.66	0.1	84	0.2
—	84.5	—	29.6	—	—	—		—	
1975	95.5	166	20.8	35.7	—	0.62		87	
6325	43.5	330	9.3	26	—	0.5		19	
1800	41	233	18.3	16.8	—	1.05		78	
2550	48	210	9.0	27.5	—	0.32		78	
2075	64.5	210	19.5	8.6	—	0.62		52.5	
825	76.5	163	41.7	40	—	1.22	below	82.2	below
963	83.8	178	41.2	35.7	—	1.7	0.1	80	0.2
1063	76.5	178	38.2	40	—	1.4		78	
863	67.5	150	26.3	32.5	—	2.1		84	
6600	49.5	370	9.5	33.5	—	0.5		8	
6550	61	288	19.0	30.3	—	0.94		15	
1200	43	144	15.5	28	—	1.8		74.4	
1200	43	—	15.3	—	—	—		—	

Plant Analysis

Ca	Mg	Mn	Zn	Fe	Total-P	Total-N
fresh weight						
5.40	121.25	1.88	2.03	11.93	147.50	2175
15.63	108.75	27.05	2.60	37.05	48.50	1994
31.20	112.50	17.53	2.70	20.95	53.50	3335
8.88	96.25	1.73	2.40	3.20	41.50	4368
3.40	225.00	5.85	2.40	7.10	96.00	1378
34.75	172.50	2.90	3.63	6.00	83.25	3806
8.05	86.80	6.80	22.63	86.00	56.00	1015
6.98	116.25	7.60	12.90	14.32	112.50	1389
24.30	238.75	4.35	3.40	6.68	78.00	1196
17.25	145.00	27.55	1.85	22.21	90.75	2393
48.75	242.50	90.50	3.23	26.85	87.00	2646
79.20	240.00	26.85	3.40	13.05	77.00	1305

TABLE 2 (cont'd)

Species	P dry weight	Ash content	K	Na
	%		mg/100 g	
Rooted:				
*Potamogeton perfoliatus***	17.46	4.21	9.50	12.08
Potamogeton pectinatus+	14.82	2.61	27.45	15.30
Myriophyllum spicatum+	11.55	2.50	45.48	10.78
Najas marina++	4.95	1.11	32.50	8.75
*Anacharis canadensis**	9.04	1.58	33.88	8.65

+ Pond Zalavári; ++ Pond Vörsi; * Zala II (canal draining Pond Vörsi); ** Zala III

plant. The aerohydatophytes accumulated ten times, the submerged plants thirty-forty times more Mn, than did the bog plant. The Zn concentrations were nearly the same, except for the rooted aerohydatophytes, where it was one order of magnitude higher than in the other plants. The Fe

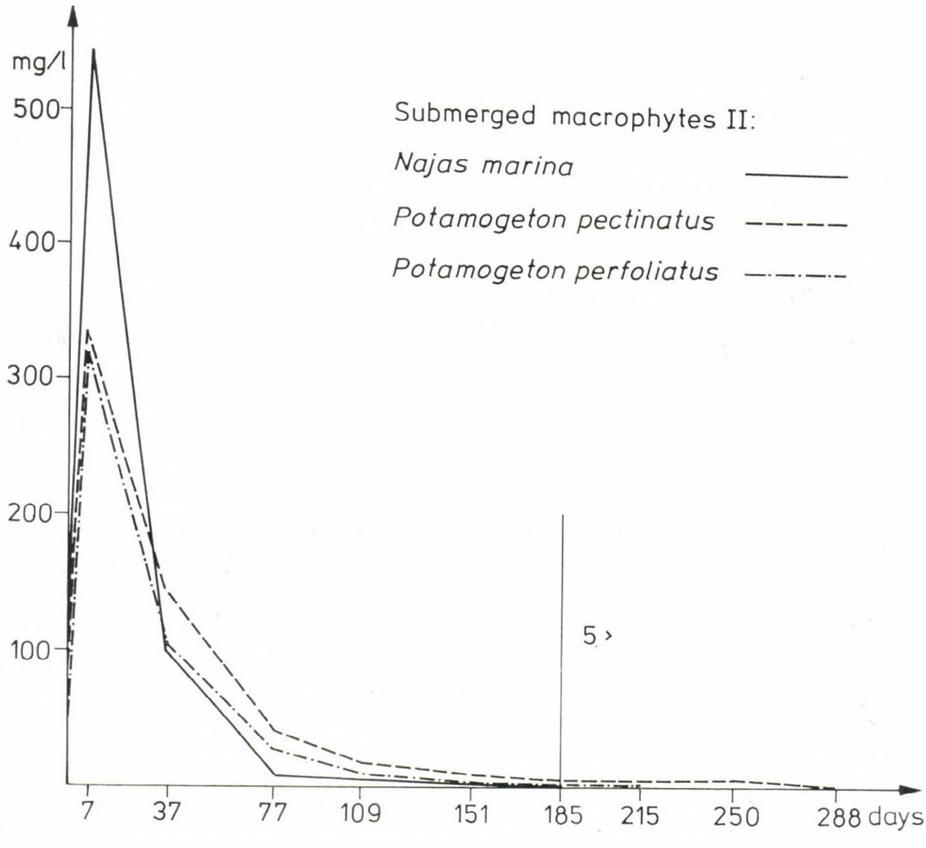


Fig. 2. K release during plant decomposition (1975–1976)

Ca	Mg	Mn	Zn	Fe	Total-P	Total-N
fresh weight						
150.50	270.00	19.10	3.60	23.00	48.00	3190
40.80	168.75	29.53	3.60	95.73	92.75	1559
89.70	159.38	55.33	2.70	16.86	100.13	1836
11.73	611.25	9.38	2.60	8.95	47.25	3154
19.58	68.75	36.50	3.25	51.50	79.75	380

+Pond Zalavári ++ Pond Vörsi; * Zala II (canal draining. Pond Vörsi); ** Zala III

content was the highest in the rooted submerged plants. Concerning the P content, the bog plant was the first. There was no significant difference in the N content of bog and water plants.

It was established that the rooted and floating aerohydatophytes accumulate nearly the same amounts of nutrients. This is explained by the structure of the stands and by the similar character of the individuals. In general, the highest accumulation has been found among the submerged plants. They contained amounts of Mn and Fe two orders of magnitude higher than the aerohydatophytes. Our results show that hydatophytes accumulate large amounts of nutrients. These studies support our suggestion put forward in earlier publications that aquatic macrophytes are of importance in water purification. To continue the logical sequence, the plants not removed from the water return their nutrients into the environment. This has made it necessary to study the nutrient release during plant decomposition. Such studies have been going on in our laboratory since 1974. The percolation experiments start a few hours after the collection of plant material. So far 17 species have been investigated.

K release (Fig. 2) showed the same tendency in all plants. The highest concentration was found in the second drainage, i.e. on the 7th day. By the 77th day more than 90 per cent of K had already been released by all the investigated plants.

Na release (Fig. 3) was similar to that of K. Here, too, the peak value was observed on the 7th day. There was a rapid decrease up to the 109th day, thereafter the decrease became slower (Fig. 3).

The release of Ca during decomposition of submerged macrophytes showed the same tendency as that of Na and K, about after the 37th day the decrease was already slow, and even after the 453rd day some Ca was retained (Fig. 4).

Mg release was rapid in the first 37 days, then it became slow (Fig. 5). Mn release was not studied due to technical reasons.

Of the micronutrients, Zn and Fe were determined. Zn concentration was maximal in the percolation fluid on the 1st day, then it gradually decreased and no Zn was found on the 25th day (Fig. 6). Fe was practically completely released from the submerged macrophytes before the 37th day, but traces were detected up to the 109th day (Fig. 7).

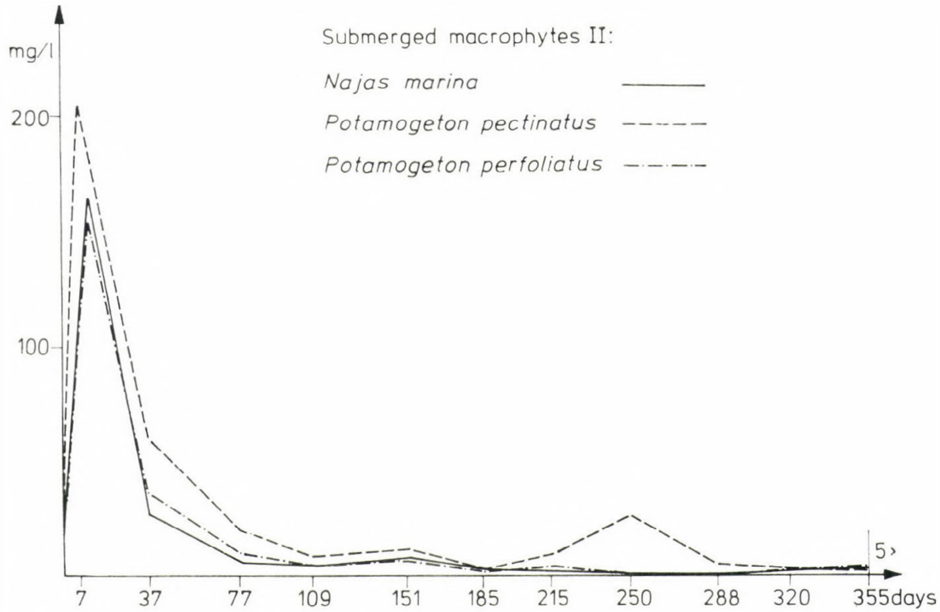


Fig. 3. Na release during plant decomposition (1975–1976)

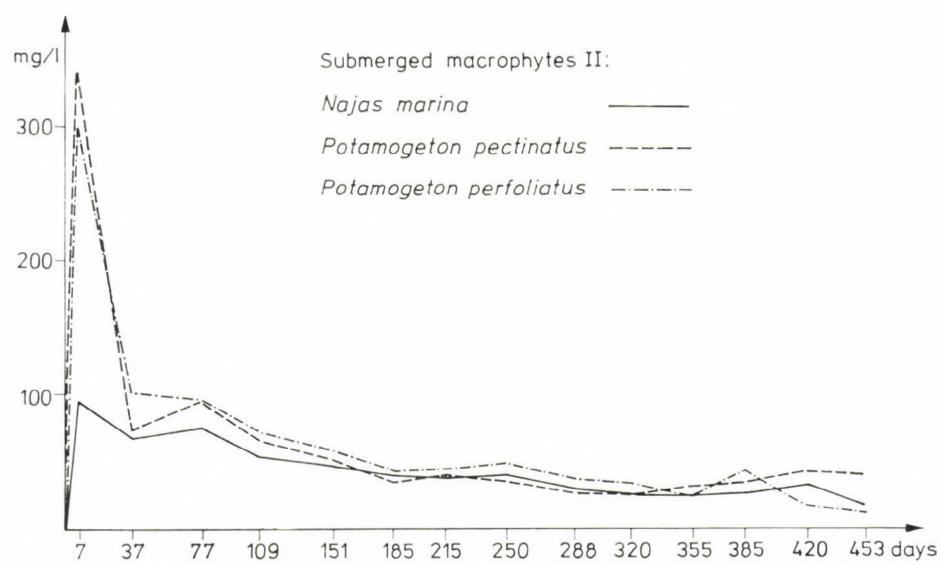


Fig. 4. Ca release during plant decomposition (1975–1976)

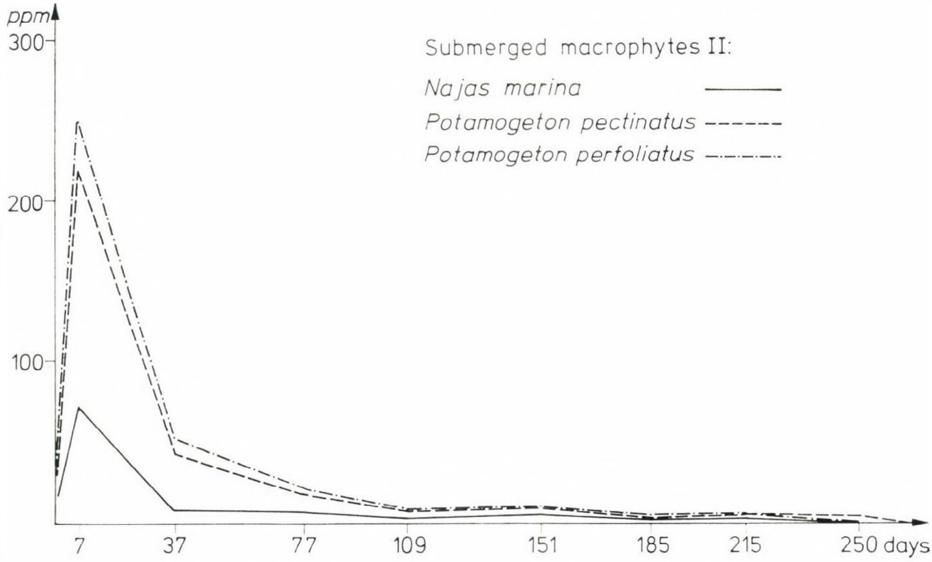


Fig. 5. Mg release during plant decomposition (1975–1976)

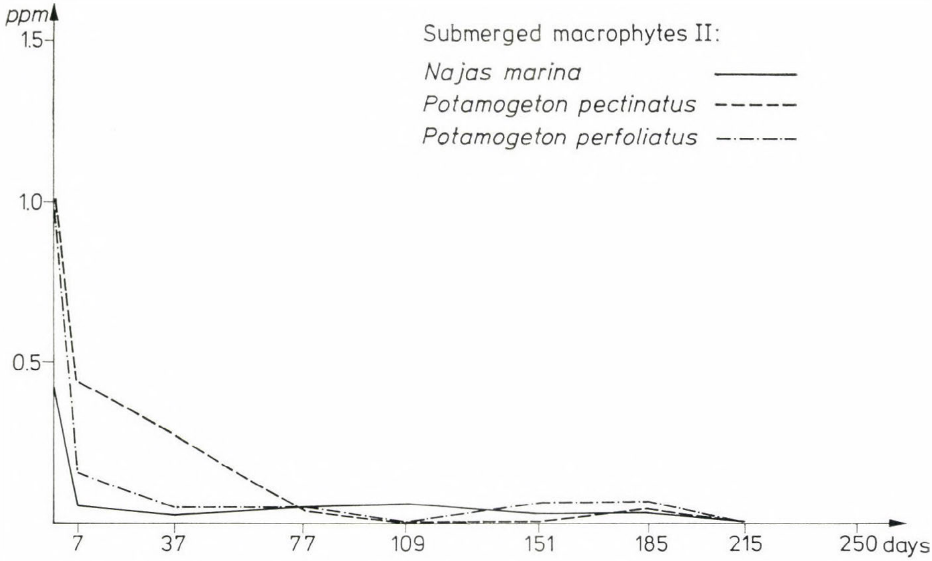


Fig. 6. Zn release during plant decomposition (1975–1976)

P and N release, due to the complex nature of the problem, is going to be dealt with in a separate paper.

As demonstrated above, the highest concentration of nearly all investigated elements was found in the percolation fluid before the 7th day, and decrease was very slow after the 37th day.

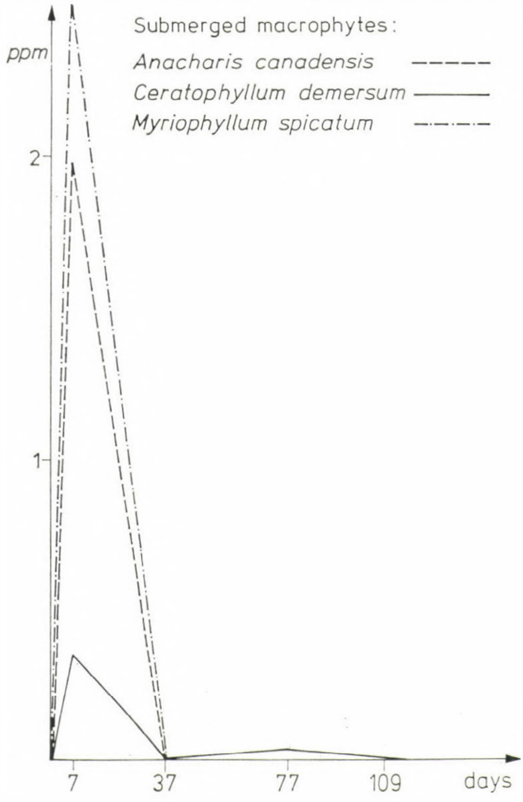


Fig. 7. Fe release during plant decomposition (1975–1976)

Using these data and the results of biomass determinations the total amount of nutrients in the given plant species in the given area will be estimated. The complex study of the nutrient content of the biotope, of the phytomass production, and that of the accumulation and release of nutrients will enable us to estimate the importance of aquatic macrophytes in the eutrophication of the lake.

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DECLINE OF REED SWAMPS IN LAKE CONSTANCE

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Abstract

Under high nutrient supply the reed belt produces high population densities and stems with reduced mechanical stability. Great amounts of filamentous algae remain entangled inside the reed community. These conditions lead to a hindered water exchange between lake and reed. The decaying organic material consumes the dissolved oxygen often until anoxic conditions appear. The breakdown process releases H_2S in toxic concentrations. The clearing away of organic matter by cutting or burning considerably reduces the organic matter content in the sediment.

The reed plant *Phragmites communis* has an ubiquitous distribution and is capable of tolerating highly different ecologic conditions. According to the studies of various research workers (Rudescu 1974, Ivlev 1950, Hürlimann 1951, Björk 1967, Klötzli 1971 and others), its optimal conditions in lake water and sediments which are well known, show a wide variety. Not so much has, however, been reported on limiting factors and their concentrations.

Although the general chemical conditions of both water and sediments in the reed area of Lake Constance are within the limits reported in the literature, a decline of the reed swamps has been observed.

1. *The extent* of the reed-covered area of Lake Constance has decreased by about 35 per cent in the recent 50 years (Grünig 1975). This could be demonstrated using aerial photographs from 1926 and recent mapping of the *Phragmitetum*. When the water level tends to sink in winter time, varying expanses of old *Phragmites*-stem remains appear between the lake and the reed belt. The breadth of the reed belt is reduced above all at the southwestern shore of the lake. Reed invading the open water has not been observed in recent years.

2. *The population density* of the reed as seen from old photographs used to be much lower than it is at present. Unfortunately, there are no counts from that time but from the study of 'calibrated' photographs and reports of old fishermen it can be estimated that 50 years ago not more than 20 stems of *Phragmites* grew on 1 m^2. Today counts from the posterior end of Lake Constance, the Untersee, show an average population density of 48 stems per m^2, some places having densities of more than 100 stems per m^2. This increase (about threefold) can be attributed to the accumulation of plant nutrients in the lake. At the same time the dissolved phosphorus has risen from a few $\mu g/l$ to about 70 $\mu g/l$ epilimnic average concentration over the years.

3. *The mechanical stability* of the reed plant seems to be reduced although the diameter of the shoots has not obviously changed (Klötzli and Züst

1973). In autumn and winter gusts of wind, snow-fall and even birds can break down large areas of reeds. The plants are pressed to the ground and do not stand up again. They form a thicket, which is partially impenetrable for water birds, hindering the water exchange between lake and reed swamp water. Agricultural fertilization experiments have revealed that the high availability of nitrogen compounds leads to a reduction of the sclerenchymatic parts in the shoots reducing the mechanical stability of Gramineae (Tobler 1943).

4. *Filamentous algae*, above all *Cladophora*, appeared in greater quantity for the first time in 1969 and 1971, due to the increasing nutrient concentration in Lake Constance. Coming to the lake surface during periods of sunshine and calm weather they drift towards the shore and the reed belt where they remain entangled amongst the reeds till they decay. When *Cladophora* is very frequent the algal masses succeed in rolling down larger parts of the reeds.

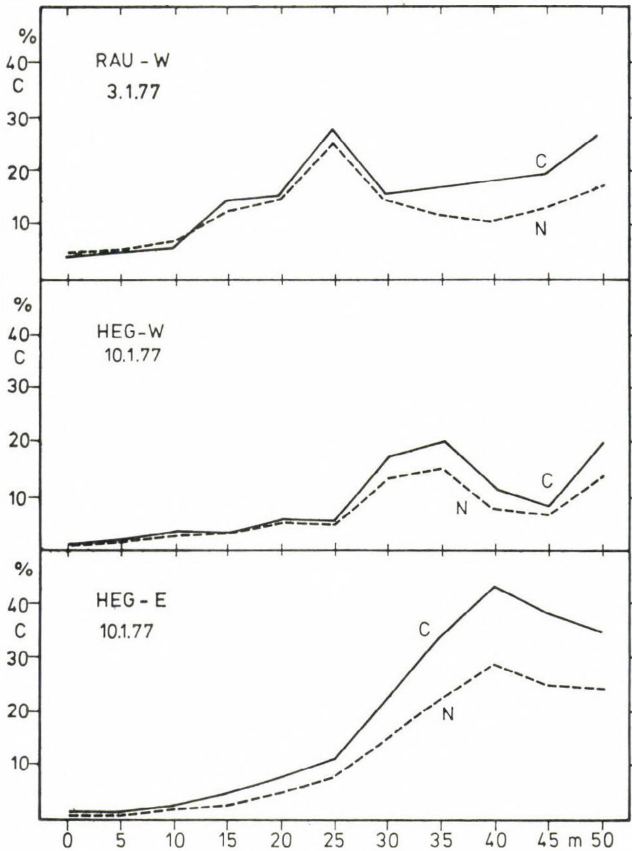

Fig. 1. Carbon and nitrogen content along three sections across the reed belt

The increase in population density, decreasing mechanical stability, and the invasion of *Cladophora* lead to a reduced water exchange between the lake and the reed swamp area. This phenomenon and the higher biomass in the reed area are the cause of the development of special chemical conditions in the sediment and water.

5. *The sediment* was described as stony, sandy and muddy, but the smell of hydrogen sulphide (H_2S) was not mentioned at the beginning of this century (Baumann 1911). Therefore, it can be assumed that the content of organic matter was lower than it is today. Recent investigations of the carbon and nitrogen contents along a section across the reed belt have shown increasing values (Fig. 1) starting at the reed border with 1 to 3 per cent carbon (of dry sediment weight) and 0.3 per cent nitrogen, reaching maximum values of up to 30 per cent C and 2.5 per cent N, 20 to 40 m inside the reed and falling again towards the land, which is uncultivated with dense vegetation. The distance of the 'muddy zone' from the water–reed border depends on the exposure and the morphometry of the shore. In most cases, large quantities of gas, part of which is hydrogen sulphide (H_2S), are released from the bottom.

6. *The water quality* inside and outside the reed does not differ greatly as long as there is a good water exchange, i.e. if there is a low reed population density, or in the vicinity of water channels. In the typical and untouched reed swamp, however, the concentrations of organic compounds rise from 4 mg/l carbon near the lake to 11 or even 20 mg/l, conductivity ranging from 250 to 290 μS, dissolved phosphorus from 70 μg/l in the lake to 240 μg/l, ammonia-nitrogen from a few μg to 450 μg/l and nitrite-nitrogen to nearly 70 μg/l. Nitrate-nitrogen becoming in the majority reduced tends to decrease in amount. The sum of inorganic nitrogen also diminishes as the denitrification process ends partially with the production of gaseous nitrogen. For phosphorus these results contradict the findings of Rudescu (1974) and other researchers, who found a phosphate impoverishment of reed waters in isolated plant stands. The difference might be caused by the fact that the sediments of Lake Constance reeds dry up and can partly freeze out in winter time. The breakdown under the ensuing aerobic conditions liberates plant nutrients which are incorporated again during the following vegetation growth period. This feedback is an additional nutrient supply for the reed belt, especially because the nutrient concentration is not diluted by lake water.

Phragmites takes its oxygen through the leaves and a small amount through adventitious roots. These roots reach down in the water to just above the bottom. It is known from the work of Rudescu (1953, 1965) that H_2S concentrations of 5 mg/l can destroy a reed population rapidly. In reed stands with reduced water exchange a metabolic system with strongly varying conditions arises. The variability depends partly on the wave action and stronger currents in the lake and also on water level fluctuations, causing an equilibrating or buffering effect, especially after stormy weather in wind-exposed areas. In such cases, many chemical factors in the water, even 50 m inside the reeds, can be the same as in the lake and some time is required before typical conditions are restored again.

When water exchange is poor, as is the case in many reed areas of Lake Constance, specific chemical conditions arise. They can be characterized by low and fluctuating oxygen tensions, high ammonia and hydrogen sulphide ion concentrations and the presence of H_2S above all in the central parts of the reeds where the carbon content in the sediments is over 25 per cent. In these places the oxygen concentration strongly depends on the time of the day (Fig. 2). It reaches a maximum in the late afternoon and a minimum just before sunrise. As it is very difficult to obtain undisturbed field measurements, we started laboratory experiments with reed sediments under controlled light conditions. The first results are as follows. The mean oxygen level depends on the relation between the dark and the light phase. When the dark period is short (in the experiment 17 hours) the oxygen fluctuations are on a higher level and the minimum concentrations do not reach zero. When the dark period is 19 hours long, the fluctuation curve is lower and the dissolved oxygen may disappear completely by the end of the dark period. Hydrogen sulphide from the bottom sediments, being constantly produced, no longer becomes oxidized. This means that critical conditions for the reed will, if ever, appear in autumn when the nights become longer. As this is also the end of the vegetative period with lower radiation, the oxygen supply from primary production also decreases. At the same time, oxygen consuming material such as *Cladophora* and macrophytic remains, gathers in the reeds worsening the conditions, too. Therefore, the critical conditions arise in September and reach their maximum at the end of October. On the other hand, at this time *Phragmites* finishes its seasonal development so that the critical condition does not fully affect the plants.

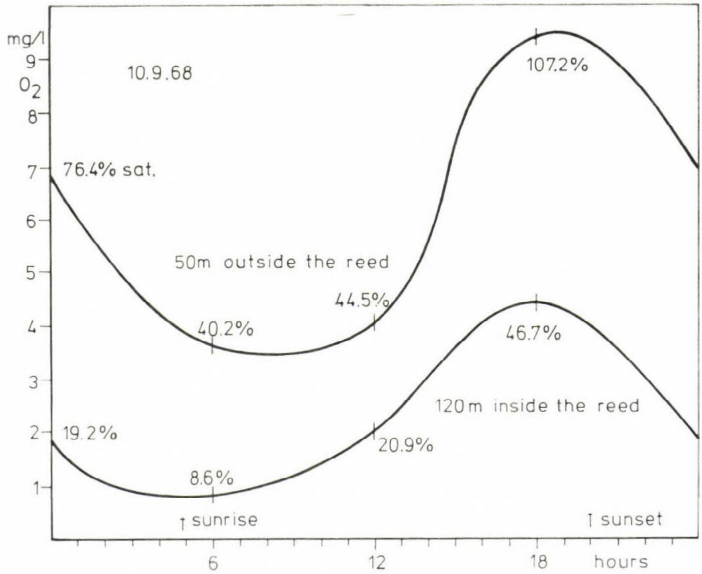

Fig. 2. Oxygen concentrations in the reed and open water over a day cycle

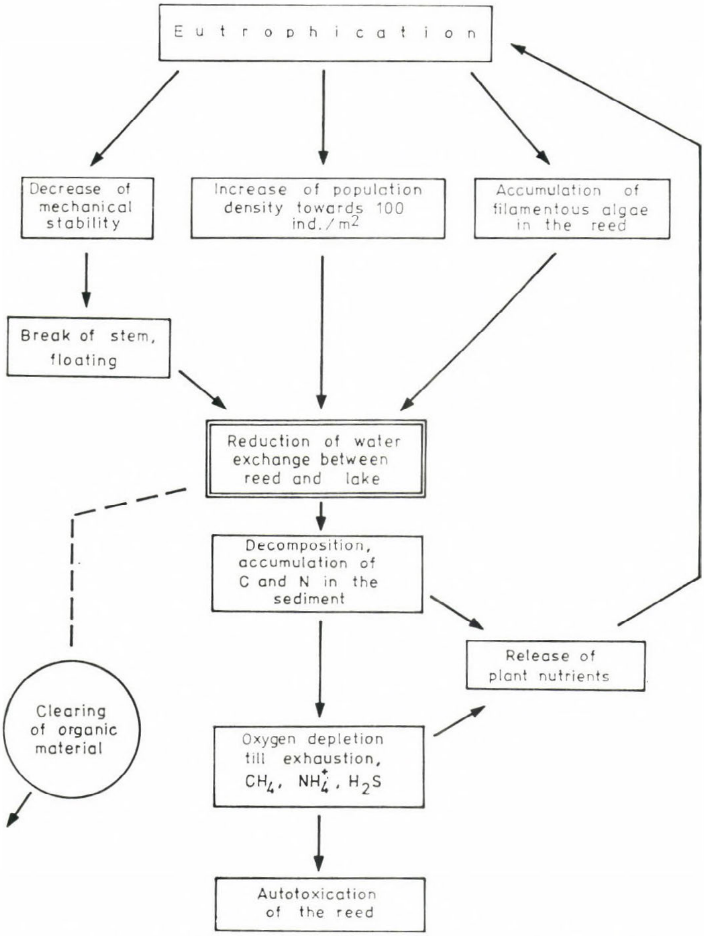

Fig. 3. Flow diagram of reed damage in Lake Constance

What is meant by 'critical condition'? *Phragmites* needs oxygen neither in the sediments nor, at least for short periods, in the water. Also sulphide does not damage the plants as long as it is present as hydrosulphide ion (HS^-). Part of the sulphide is present as undissociated hydrogen sulphide (H_2S) and the ratio between the two forms depends on the pH. The concentrations are approximately equal at pH 7, but a fall of one pH-unit causes a tenfold increase in H_2S (Hutchinson 1957). In the reeds, the pH varies between 7 and 8.5, and only once was a minimum value of 6.8 measured (Banoub 1975). Therefore, the situation becomes critical for *Phragmites* when a strong oxygen depletion appears, giving way to the formation of sulphide together with a low pH. In such cases, high concentrations of H_2S may arise, mostly for some hours in the early morning, disappearing quickly with the increase in oxygen concentration after sunrise. Therefore, the chemical conditions of the reed water can be described only by recording the whole-day cycle with special regard to the critical oxygen and H_2S values.

A survey of the actual situation in the reed swamps in Lake Constance is given in Fig. 3. The flow diagram indicates possible causal connections which are neither complete nor have been quantified as yet. The eutrophication of the lake has three direct consequences: a decrease in mechanical stability, an increase in population density, and the appearance of filamentous algae. They all lead to a reduced water exchange and an additional amount of organic matter in the reeds. The decaying material consumes more oxygen than contained by the shallow water and oxygen depletion follows. If it reaches zero, phosphorus is released from the sediments intensifying the nutrient supply. The decomposition of organic matter, similarly to other substances, produces sulphides, which are present partly as undissociated hydrogen sulphide (H_2S). Hydrogen sulphide, especially if the pH is near 7, or less, leads to an autotoxic effect in *Phragmites*.

To stop this unfavourable development two methods have been tried to reduce the organic load: burning or cutting of one reed generation in wintertime when the material is dry. Both methods have nearly the same effect. The plants of the new generation are a little shorter (by about 10 per cent) and their stems a little larger in diameter (by about 19 per cent). The population density seems not to be reduced significantly. The organic content in the sediments, however, decreases by 40 per cent above all at places inside the reed swamp with high organic load. These results have to be confirmed at several different localities before a cutting program can be established.

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ALGAL VEGETATION OF LAKE NEUSIEDL AND ITS NATURAL AND MAN-INDUCED CHANGES

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Abstract

Lake Neusiedl shows considerable fluctuations in its salt content resulting from its changeable water level. Long-term alterations in the proportion of the individual ions have been detected. Formerly, Lake Neusiedl was regarded as a soda lake, but recent research has shown that Ca^{2+}, Mg^{2+} and SO_4^{2-} have greatly increased in the northern part.

The unique algal flora of this lake has not been adequately studied so far. Since about 1963 samples of whole algal biocenoses (benthos, plankton) have been studied regularly. During the last 10–15 years great floristic and quantitative changes have been observed: some species, formerly rare, suddenly came to dominance (*Bacillaria paradoxa, Botryococcus braunii*). Other species have vanished gradually and have been replaced by new ones. Tourism, industrial development, and destruction of the reed belts have caused increasing eutrophication. Following this, chlorophytes have developed luxuriantly. *Pediastrum duplex*, not at all recorded by Loub in 1955, dominated the plankton in 1975. Showing a similar appearance are *Oocystis lacustris, Planktosphaeria* and *Sphaerocystis*. In 1975–1976 another new alga appeared in great numbers: *Lobocystis dichotoma*, never found in Austria before.

The origin of these species and the circumstances of their mass growth are completely unexplored. A new diatom, *Gyrosigma macrum*, may have been transported from the marine coast with young eels from the North Sea. Further studies in taxonomy and nutriments of algae of Lake Neusiedl are needed.

Lake Neusiedl, situated on the margin of the Western Hungarian Plain, within the area of influence of the Pannonian climate, extends about 32 km in north-southern and 4–10 km in east-western direction. The lake covers about 300 km² but it is only 0.5–2 m deep (since 1965 maintained at 1.75 m according to an agreement between Austria and Hungary) and 50 per cent of its area is covered by reeds (*Phragmites communis* Trin.). Due to the prevailing winds (mostly of north-western direction), the loose mud is frequently stirred up from the bottom greatly increasing the turbidity of the water. Therefore the degree of light penetration is not very high, and Secchi-transparency is less than 10 cm (Sauberer 1952, 1953). This has a great influence on primary production (Dokulil 1973). Owing to the shallow depth, the temperature of the lake follows that of the air very fast: it may reach 25–30 °C within a few days in summer, and ice covers the lake for 10–97 days in winter.

Beside these morphological and physical particularities, the chemical composition of water, too, differs from that of other lakes. In long-term alterations the total salt content changes along with water level fluctuations (Stehlik 1972). In 1937 the sum of all ions was 3.5–16 g per l, and the lake was 50–70 cm deep. By contrast, in 1957 the sum of all ions was 1.8 g per l, and water depth was 80–100 cm.

The ratio of individual ions, too, changed concurrently with fluctuations in the water level. In the last ten years the ratio of Ca^{2+}/Mg^{2+} to Na^{+} has changed to a great extent. Formerly Lake Neusiedl was regarded as a soda lake, but recent results (Neuhuber 1971, Stehlik 1972) have shown that nowadays this is true only of the southern part, whereas Ca^{2+}, Mg^{2+} and SO_4^{2-} have greatly increased in the northern section. This presumably occurs as a result of influx of interstitial waters rich in Ca^{2+} and Mg^{2+} ions. Fountains with high sodium content predominate in the southern part, but the drainage of these waters is considerable to the south-east via the artificial Einser Canal (since 1910).

Owing to the flat margins of the lake, a drop in the water level enlarges the marginal areas where salt concentration may rise considerably by evaporation. Heavy rains decrease the salinity to some extent.

The algae of Lake Neusiedl have not been sufficiently studied so far. Data on algae (about 50 diatom species) found in Lake Neusiedl before the complete desiccation of 1865 were listed by Grunow (1860–1863). Pantocsek (1912) also studied the diatoms and enumerated 149 species, but several of them were synonymous. Further publications on the algae of Lake Neusiedl are due to Loub (1955), Schiller (1955–1957), Hustedt (1959*a*, *b*), Schmid (1973), Kusel-Fetzmann (1974). Studies dealing with selected species and their development, sexual reproduction etc., were presented by Geitler (1969, 1970*a*, *b*) and Tschermak-Woess (1972, 1973*a*, *b*).

Since 1963 I have been taking samples from many locations of the open lake and the reed belt: periphyton, benthic algae and plankton, for purposes of research on algal vegetation. A preliminary study was published in 1974, dealing with chlorophytes and desmids. Current studies are focussed on euglenophytes.

ALGAL COMMUNITIES

In the open lake, algae cannot be clearly distinguished in pelagic and benthic forms. The difference between the open lake and the various habitats in the reed belt is greater. In the open lake the turbulence causes plankton samples to contain a mixture of true planktonic forms (thin *Nitzschia acicularoides* Hust., *Monoraphidium*, *Euglena acus*, *Botryococcus braunii*) and large-sized benthic forms like *Surirella peisonis* and *Campylodiscus*. Very characteristic are the long *Euglena oxyuris*, *Euglena tripteris*, *Euglena ehrenbergii* in this habitat. Great changes of species composition have been noticed in the plankton recently. Ruttner-Kolisko and Ruttner (1959) have reported *Monoraphidium* (= *Ankistrodesmus falcatus* var. *spirilliformis*) dominating with up to 3 million cells per l. This species receded later on, exceeding, however, these values again in 1972. *Oocystis lacustris* appeared for the first time between 1968 and 1970. *Botryococcus braunii* was predominant in 1970, especially in the southern part. *Pediastrum duplex* not found in the lake earlier by either Loub (1955) or myself, predominated in 1975. *Sphaerocystis* and *Planktosphaeria gelatinosa* appeared simultaneously. Another chlorococcal alga appeared in Lake Neusiedl in 1975: *Lobocystis dichotoma* (Fig. 1), which has been described by Thompson (1952) in the United States and has never been reported in Lake Neusiedl in Austria until this finding.

Benthic biocenoses can develop in the open lake only in winter beneath the ice or, else, in calm bays or near the reed belt. These biocenoses consist of diatoms (*Gyrosigma*, *Amphiprora costata*, *Nitzschia* and *Navicula* species, *Surirella peisonis*), chlorophytes, euglenophytes and frequently of blue-green algae (*Oscillatoria*, *Spirulina*). Such epipelic layers may be detached from the bottom by gas bubbles originating from photosynthesis, drift to the water surface, scattering their constituents to the plankton.

In the canals and the open parts inside the reed belt, the water is clear but brownish in colour. This is caused by the destruction of organic matter

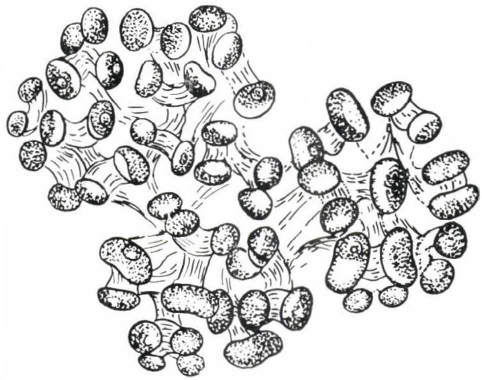

Fig. 1. Lobocystis dichotoma

Fig. 2. Epipelic diatoms in the reed belt: *Navicula oblonga; Cylindrotheca gracilis* (↑)

releasing humic substances. The pH of between 8 and 10 in the open lake is decreased to 7.5 by higher concentration of carbonic acid. The bottom is covered with olive-brown mats consisting of algae. This biocenosis shows the greatest diversity of species. The predominances may vary over short distances and within a short time. This place is inhabited by the characteristic *Caloneis permagna*, *Anomoeoneis sphaerophora*, *Gyrosigma spencerii*, *Nitzschia lorenziana*, *Scoliotropis peisonis*, *Amphiprora*, *Navicula pygmaea*, *Cylindrotheca gracilis* (Fig. 2). *Bacillaria paradoxa*, not found by Grunow (1862), rarely seen by Pantocsek (1912), not found in samples from 1913 and 1927, was found to dominate in the forties (Hustedt 1959*a*, *b*), but was seldom found by us in the fifties. Obviously, its frequency has increased since 1965 in all biotopes. *Gyrosigma macrum*, usually known from brackish water, was found in Lake Neusiedl in these benthic communities for the first time in 1968 (Fig. 3).

The mud below this layer of living algae often turns into blackish sapropel especially in winter under ice. Then the anaerobic conditions will destroy most of the algae with the exception of H_2S-resistant forms, such as sulphur bacteria, *Oscillatoria chlorina*, *Navicula oblonga* and *Navicula radiosa*.

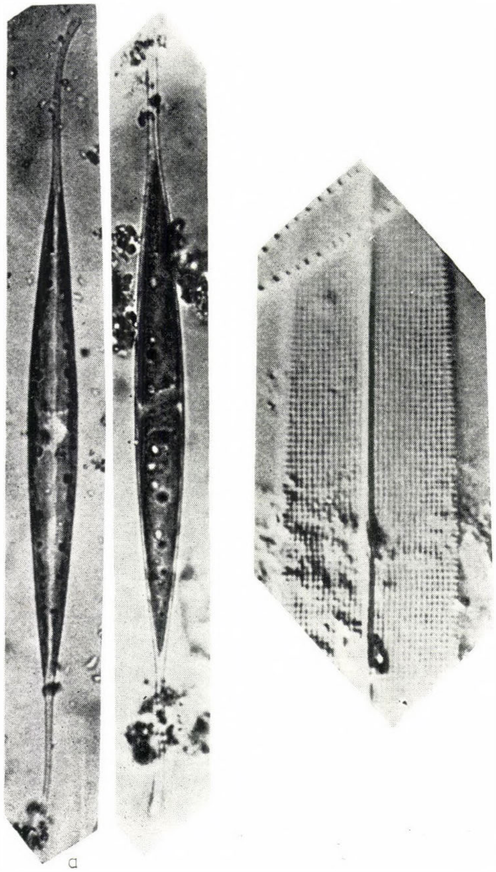

Fig. 3. Gyrosigma macrum (W. Smith) Griff. et Henfr.

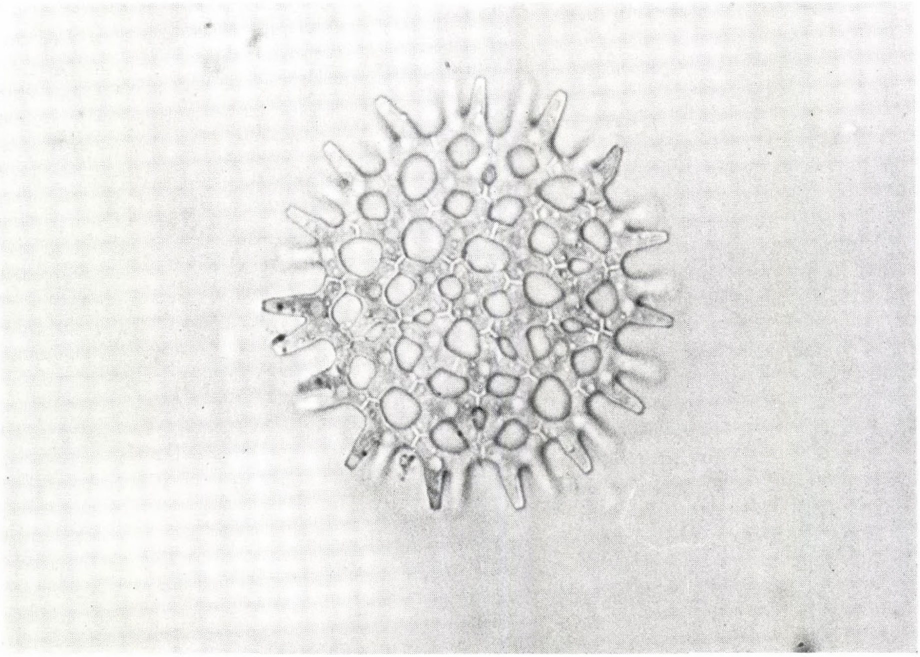

Fig. 4. Pediastrum duplex, from plankton

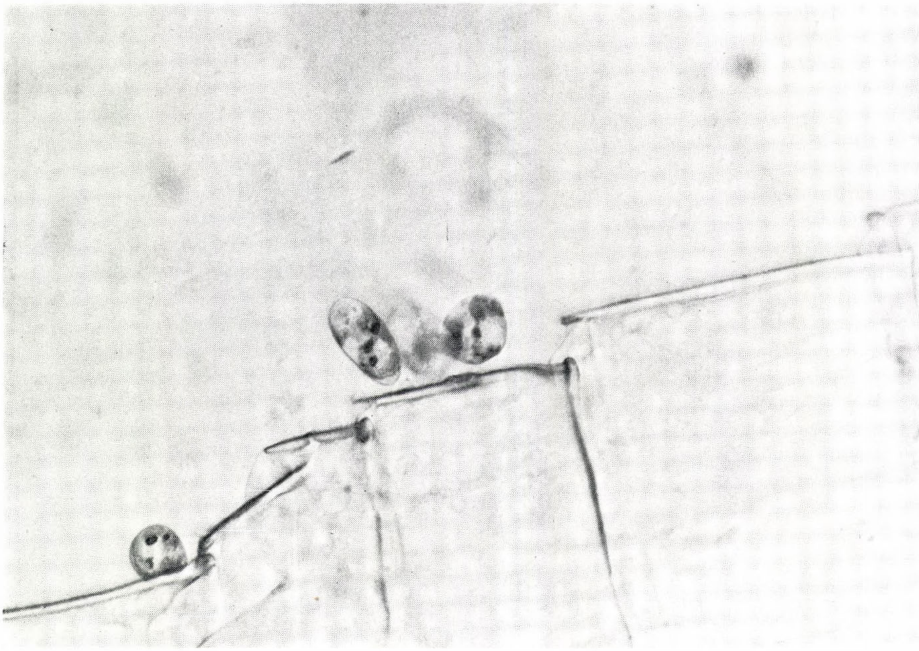

Fig. 5. Colacium cyclopicola on *Diaptomus* sp., plankton

Fig. 6. Euglena acus in metabolic bending. *Phacus acuminatus.* Canal of Weiden in the reed belt

Fig. 7. Euglena tripteris, Phacus oscillans. Canal of Weiden

Fig. 8. Phacus pleuronectes. Canal of Weiden

Fig. 9. Phacus tortus. Canal of Weiden

Periphyton communities make an important contribution to the algal flora of Lake Neusiedl. A thick gray layer frequently covers the leaves and stems of submerged macrophytes. This layer consists of mud and harbours a rich algal flora, similar to that of the bottom. Beside the diatoms, *Euglena* species, chlorococcales and desmids appear again (Figs 4–9). It is quite surprising, how many desmids are living in Lake Neusiedl, e.g. *Cosmarium biretum* Bréb., also found in the more concentrated salt pools in the Seewinkel (Stinker, Darscho) east of the lake. Within the reed belt *Utricularia vulgaris* L. forms great floating masses, also covered with thick algal mats. The reed stems themselves bear diatoms and other algae, and are often covered by *Spirogyra* and *Oedogonium* in spring. Between the threads many free-living forms are to be found. The primary production of the periphyton on reed stems has been recently studied by Sommer (1977).

Along the border line of the open water, where stronger waves attack the algae, diatoms with gelatinous stalks (*Gomphonema olivaceum* var. *calcarea*, or *Cymbella*), or even living within gelatinous tubes (*Cymbella lacustris*, *Nitzschia filiformis* and *Amphipleura rutilans*) will occur. According to Drum (1969) and Schmid (1973) such gelatinous tubes provide the diatom cells living inside with a milieu independent of the extreme water chemistry. It is interesting that such forms are also found in the extreme soda lake Van-göl in Turkey.

The most recently studied group of algae of Lake Neusiedl are the diatoms, but we do not know enough about them. Hustedt (1959*a*, *b*) reckons that about 150 species exist in the area. This is a small number compared with the number of species in other waters. Some species are endemic in the lake or are restricted to natron lakes in Hungary (Szemes 1959) or in Persia (Kurusch-göl, Löffler 1956). Such 'natron-species' are *Amphiprora costata*, *Caloneis permagna*, *Surirella peisonis*, *Scoliotropis peisonis*, *Cylindrotheca gracilis*.

Resistant eurytopic species, otherwise inhabiting fresh or slightly polluted waters, are the most typical components of diatoms (*Nitzschia sigmoidea*, *Cymatopleura solea*, *Caloneis amphisbaena*).

A third group originates from the marine and brackish waters of the sea coast. To these belong *Bacillaria paradoxa*, *Gyrosigma macrum* and *Nitzschia navicularis*. Schmid (1973) was able to establish the origin of a number of species from Lake Neusiedl by proving their osmotic resistance against the ions Na^+, Mg^{2+}, Cl^-, SO_4^{2-}, CO_3^{2-}. No other algae have so far been proved to have such a resistance.

The origin of new algae appearing in the lake is unknown. As to the diatom *Gyrosigma macrum*, they were probably introduced into Lake Neusiedl with young eels, caught at the coasts of the North Sea (Kusel-Fetzmann 1973). *Gyrosigma macrum* is a diatom living in the sandy banks of brackish waters. But as far as the chlorophytes *Sphaerocystis* or *Planktosphaeria* are concerned they have possibly been brought in accidentally by boats of tourists, the samplers and nets of limnologists or by water birds.

The change of chemical conditions in the water, partly due to the natural development of the lake, partly caused by man, may alter the competitive power of some species. Tourism has been growing rapidly in recent years, new settlements have been built in the reed and large portions of the reed belt have been destroyed. Thus the cleaning power of the reed belt decreases, waste water, formerly filtered by the reed, reaches the open lake: the eutrophication causes an explosion of chlorophytes. The only greater influx, the Wulka, is heavily polluted. All this has resulted in an increase of the phosphorus content of the lake during recent years, as shown by Neuhuber (personal communication). Further studies are needed in the taxonomy of algae as well as regarding their nutrition.

The changes of algal species, and their explosion-like growth point to an increasing pollution and must be regarded as signs to stop this development and to take action to protect the lake against eutrophication. A very important prerequisite for preserving the purity of Lake Neusiedl is the preservation of an intact reed belt.

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CHANGES IN THE PHYTOPLANKTON–ZOOPLANKTON RELATIONSHIP CONNECTED WITH THE EUTROPHICATION OF LAKES

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Abstract

Some results of long-term studies on an eutrophic, lowland lake (Lake Mikołajskie) undergoing cultural eutrophication are presented. The effect of large-sized phytoplankton (e.g. the dinoflagellate *Ceratium hirundinella* and blue-green algae) on the composition and quantitative development of crustacean plankton in stratified lakes at different levels of eutrophication in the different seasons is shown. The question of efficiency of organic matter transformation in pelagic food relations is discussed.

INTRODUCTION

The primary effect of eutrophication in the pelagic zone of temperate lakes originates from the trend of an increase in algal biomass usually observed in successive years, particularly in summer. The increase of algal biomass and the longer duration of algal blooms provide new trophic conditions for primary consumers, i.e. for suspension-feeders, like planktonic cladocerans, calanoids and rotifers. This interaction is first of all a result of changes in the absolute abundance, size structure, nutritive value and assimilation of suspended food particles. The size and structure of food suspension will be different when the heavy algal growth induced by eutrophication involves species of small cell size directly available for filter-feeders (accessible for their filtering apparatus) or when large cells or colonies (like blue-green algae) dominate within the algal biomass.

In the latter case only the decomposition products of algae, bacterial cells and fine detritus will be consumed by suspension-feeders directly. The bacteria and detritus particles are usually of a smaller size (a few μm) in food suspension in comparison with the cells of nannoplanktonic algae accessible for filter-feeders (up to 20–30 μm). For this reason the relative abundance of these two components in total food suspension may regulate the species composition of filter-feeders and stimulate the growth of fine filtrators such as some cladocerans and small rotifers (with a preference for smaller particles), or coarse filtrators, like calanoids (with a preference for a wider spectrum of particle sizes). The increase of algal biomass induced by eutrophication should be followed by appropriate quantitative and qualitative changes in the primary consumers.

However, looking for a logical explanation of long-term changes in the zooplankton caused by eutrophication, the effect of various 'masking' factors is evident. Disregarding the influence of worsening oxygen conditions which, in themselves, may affect the occurrence of zooplankton, the following factors may be of interest as far as trophic relations are concerned.

1. Changes in the composition of fish fauna feeding on zooplankton. The influence of fish predation on zooplankton composition is well known, so changes in the ichthyofauna resulting from eutrophication, as e.g. the disappearance of coregonids (Larkin and Northcote 1969) may affect the composition of zooplankton. The observed long-term changes in the zooplankton connected with eutrophication can, according to Brooks (1969) be attributed to changes in the ichthyofauna rather than to the direct influence of phytoplankton. Probably, similar effects will be produced by invertebrate predators. Gliwicz et al. (1977) have shown that the planktonic predators (Cyclopidae, *Leptodora kindtii*) can eliminate quite an important part of the total zooplankton biomass.

2. Input of allochthonous organic matter from various sources (soil erosion, sewage, littoral debris, etc.) to the pelagic zone. This matter can stimulate bacterial and detritus production independently of the autochthonous sources and thus influence the size, structure and abundance of food particles.

3. Finally, many factors directly or indirectly connected with the climatic differences of successive vegetative seasons should be taken into account, which may affect the timing of biological cycles, the conditions for survival of resting stages, etc., and, accordingly, might affect the occurrence of species quite independently of the 'eutrophication' factors.

In the present paper it will be attempted (*i*) to interpret some long-term changes in the phytoplankton and zooplankton observed in a typical eutrophic lowland lake, i.e. Lake Mikołajskie, and (*ii*) to compare the data of lakes with different phytoplankton biomass in the summer period.

GENERAL CHARACTERISTICS OF LAKE MIKOŁAJSKIE

Lake Mikołajskie [53° 47' N latitude, 21° 36 E longitude, 116.1 m a.s.l., $\tilde{z}$ 11.0 m, z_{max} 27.8 m, volume 50.5×10^6 m^3, surface area 460 ha, development of shore line (SD) 1.7, area of the littoral 19%] is a typical eutrophic, post-glacial lake where physicochemical parameters and biota were studied from 1963 up to 1973 with different intensity in successive years.* According to Kajak et al. (1972), the amount of allochthonous organic matter loaded with run-off, shore erosion, sewage and litter fallout equals about 5 per cent of the net autochthonous production. The data of Górski and Rybak (1974) indicate that the loading rate of phosphorus, which might be taken as the maximum for the 1963–1973 period, does not exceed 0.35 g per m^2 per year. However, according to the classification of Vollenweider (1971), for lakes of $\tilde{z} = 10$ m, this value is three times that of the permissible level (0.1 g/m^2 per year). Urban sewage is the main source of phosphorus input to the lake. Sudden changes in lake nutrient concentrations were, however, not detectable in the study period (see below). This can be attributed to two factors: the small retention time of the lake water (the whole volume of the lake is exchanged three times a year), and the fact that the direct influence of sewage is limited to a small area.

* Methods and sampling techniques are described in Rybak (1975), Spodniewska et al. (1973), Spodniewska (1974)

Chemical data (Rybak 1972, 1975) show that from 1967 to 1972, the early spring (March) concentrations of nutrients did not exceed 0.03 mg per 1 for inorganic phosphorus and 0.2 mg per 1 for inorganic nitrogen. These values did not show any clear trend in the study period but rather strong fluctuations from year to year. The oxygen and pH values were similarly fluctuating. A small increase of conductivity and calcium concentration could be recorded in 1970 as compared with 1967.

The lake has a drainage area of 810 ha, 30 per cent of which is covered with forests and 60 per cent utilized by agriculture (half of the area are meadows). In the warm periods, the region is a frequented holiday resort with about 15 000 tourists visiting daily the town of 3000 inhabitants situated on the lake's shore. The raw sewage in an amount of roughly 220 tons dry weight per year is disposed to the lake (Górski and Rybak 1974).

According to the classification of lakes based on fish fauna, Lake Mikołajskie is of a vendace (*Coregonus albula* L.) type where the natural population of vendace, enriched every year by the introduction of fry, was maintained during the whole study period, although, judging by the catches, at a fluctuating level. Total catches during the study period varied considerably from 150 to 50 kg per ha, without the changes showing any clear trend. Plankton-feeders like bleak, smelt and vendace dominated in the catches of pelagic fish, while bream dominated in the total catches.

On the basis of the available data, it can be assumed that during this period of ten years Lake Mikołajskie remained under the rather moderate influence of eutrophying agents, which did not drastically change the physicochemical characteristics of the lake. It should be emphasized, however, that this statement holds true only for the period discussed, since random sampling in recent years has shown a sudden deterioration of water quality (e.g. in summer 1977 water transparency was less than 1 m, and a high level of oxygen deficit occurred).

LONG-TERM CHANGES IN PHYTOPLANKTON AND ZOOPLANKTON IN LAKE MIKOŁAJSKIE

Although the eutrophic situation in Lake Mikołajskie was not alarming in the analysed period, significant changes of phytoplankton abundance and composition during the period 1963–1972 were documented by Spodniewska et al. (1973) and Spodniewska (1974, 1976).

Owing to the seasonal differences observed in these changes, two periods have been distinguished: (*i*) the spring season (second half of April and May) characterized by total water circulation (average temperature of the water column at the end of circulation was about 10 °C) and domination of diatoms in the phytoplankton; and (*ii*) the summer stagnation season (August) with sharp temperature and oxygen stratification (average temperature in the epilimnion about 18 °C, in the metalimnion between 12 and 8 °C, and under 10 °C in the hypolimnion) accompanied by domination of dinoflagellates in the phytoplankton.

The nannoplanktonic algae up to 30 μm in size were distinguished in the phytoplankton in both periods. They were mostly flagellates and diatoms. According to Gliwicz (1969, 1974), this size ($\leq 30\ \mu$m) covers the range of food particles (up to 20 μm) available to most suspension-feeders common in temperate lakes.

Analysis of long-term changes in the phytoplankton in the years 1963–1972 (Fig. 1) clearly reveals that (*i*) in spring the total phytoplankton biomass is maintained at a more or less steady level of about 10 mg per l fresh weight, while the nannoplankton biomass shows a rising trend, which causes an increase of its share in the total biomass (Fig. 2). (*ii*) In summer, on the other hand, the total phytoplankton biomass shows a strongly rising trend in the surface layer (Fig. 1), particularly when the period 1969–1972 is compared with earlier years; the respective values are 10 and 30 mg per l. A dinoflagellate, *Ceratium hirundinella* (O.F.M.) Bergh, is responsible for this increase of the biomass (see Fig. 2). Its share in the summer biomass increased during 1969–1972 from 60 to 90 per cent replacing nannoplankton and blue-green algae [*Oscillatoria* sp. div., *Anabaena* sp. div., *Aphanizomenon flos-aquae* (L.) Ralfs, and *Gloeotrichia echinulata* (J. S. Smith) Richt.]. In this period, the nannoplankton biomass showed a clearly decreasing tendency. The period of *Ceratium hirundinella* bloom, too, increased, from 20 to 30 days in the years 1963–1964 (from the middle of July to the middle of August) and to almost 80 days in the years 1970–1972 (from the beginning of July up to the end of September).

Based on the above data, the following conclusions can be drawn. First of all, the increase of the biomass of dinoflagellates and not of the blue-greens, as commonly observed elsewhere, was a symptom of eutrophication in Lake Mikołajskie in the decade analysed. At the moment it is rather difficult to find the possible causes of this phenomenon. However, such a phenomenon appears to be quite common in deeper lakes of Northern Europe (Scandinavian countries, Great Britain). This is probably the first stage of eutrophication in deep, stratified lakes, brought about by moderate pollution. In addition, the blue-greens which occurred in greater numbers in the spring and summer of the years 1963–1967 (from 0.1 to about 2 mg per l) became less abundant in the corresponding seasons of 1969 and 1972 (0.1 to 0.5 mg per l, respectively). However, it should be emphasized that these data apply only to the decade between 1963 and 1973, which may represent a stage of moderate eutrophication in the life of Lake Mikołajskie. Recent unpublished data show irregular increases of the biomass of blue-green algae (mostly species of the genera *Oscillatoria* and *Anabaena*).

Secondly, the changes in the algal flora in the spring and summer periods must have had different effects on the zooplankton. In the spring periods, the abundance of consumable small-sized algae increased, while in the summer periods, the abundance of *Ceratium hirundinella* with large cell size increased (above 200 μm). This species is not accessible *in vivo* to suspension-feeders. The bulk of phytoplankton in summer is thus only available for filter-feeders after bacterial decomposition.

A direct evidence that the decomposition rate had increased in successive summer periods was reported by Spodniewska (1974). The absolute values of daily oxygen consumption in the trophogenic layer as well as the relative values (as a percentage of gross primary production) showed a rising trend during the period of 1967–1972 (70 to 97 per cent, respectively). Another piece of evidence is furnished by the data of Godlewska-Lipowa (1975) who observed an increase of bacterial biomass in the surface water from 1971

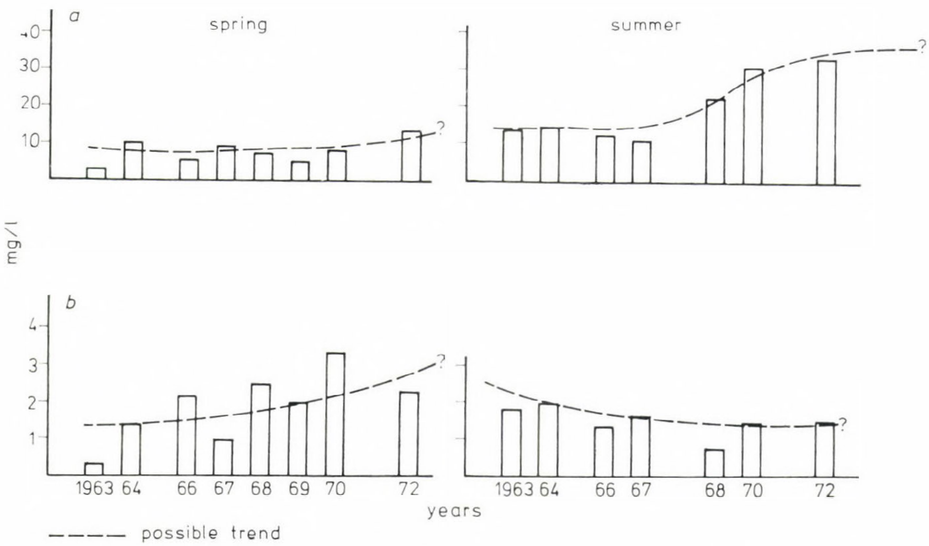

Fig. 1. Average biomass of total phytoplankton (*a*) and nannoplankton (*b*) in two seasons of different years (Lake Mikołajskie)

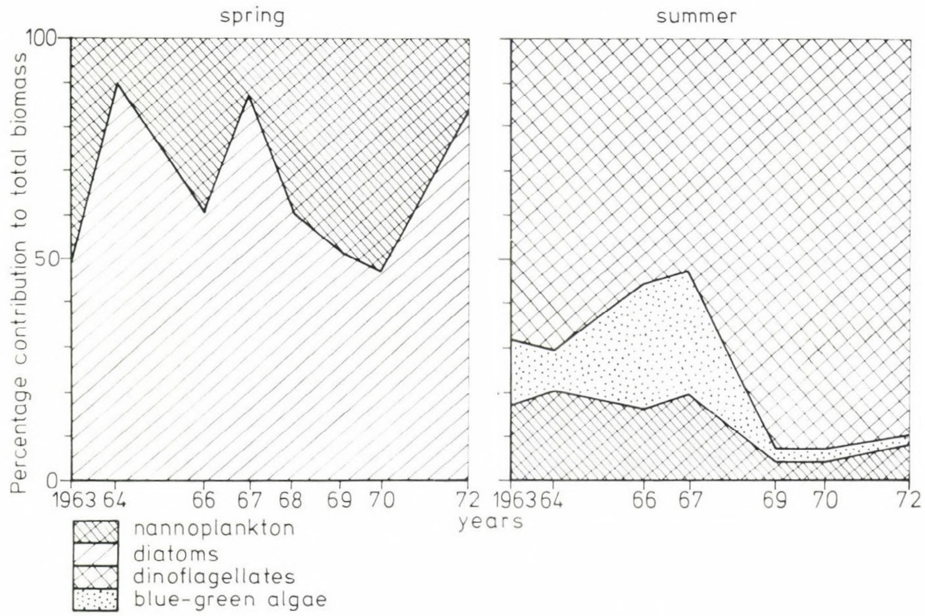

Fig. 2. Changes in the phytoplankton composition in two seasons of different years (Lake Mikołajskie)

to 1973 as compared with the period between 1969 and 1970 (corresponding values 1.38 mg per l, and 1.14 mg per l fresh weight, respectively). Although an increase was observed both during spring and summer, the greatest increment, 30–50 per cent, occurred in September and October, i.e. during or after the peak of water bloom caused by dinoflagellates.

Available mean data for zooplankton for various seasons and years show considerable fluctuations from year to year, most likely caused by the variability of biological cycles and by changing climatic conditions. Nevertheless, certain tendencies can be distinguished (dotted lines in Figs 3–6) and may be interpreted from the point of view of phytoplankton–zooplankton food interactions.

Attention is called to the increasing abundance of rotifers in spring (May, and the first half of June) mainly of the two genera *Polyarthra* and *Synchaeta* (Fig. 3). These rotifers prefer to feed on small flagellates (Edmondson 1965, Pourriot 1965), though *Synchaeta* is particularly successful in catching

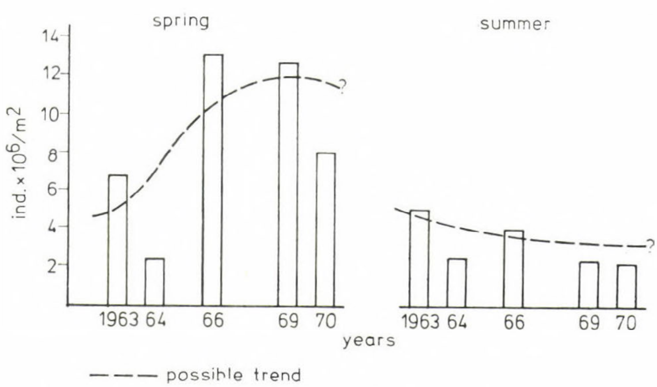

Fig. 3. Average abundance of rotifers per 1 m² in two seasons of different years (Lake Mikołjskie)

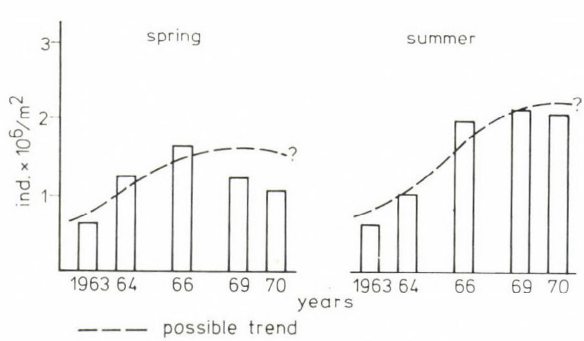

Fig. 4. Average abundance per 1 m² of crustaceans i n two seasons of different years (Lake Mikołajskie)

larger cells, too. It seems that the population growth of rotifers in successive spring periods is a reaction to the enrichment of nannoplankton cells as available food. No similar trend could be observed in summer (August). On the contrary, a decreasing abundance of rotifers was noticed in this season. The detritus-feeder *Keratella cochlearis* was continuously a dominant species.

It should be added that a growing tendency in the number of rotifers was also found in the autumn periods (September, October) as reported by Spodniewska et al. (1973) who compared data from 1966 with those from 1963 and 1964.

An increase in the number (Fig. 4) and biomass (Fig. 5) of crustaceans was observed both in spring (May) and summer (August). This increase was even more obvious in summer than in spring and involved both filter-feeding cladocerans and calanoids* in addition to predatory cyclopoids.**

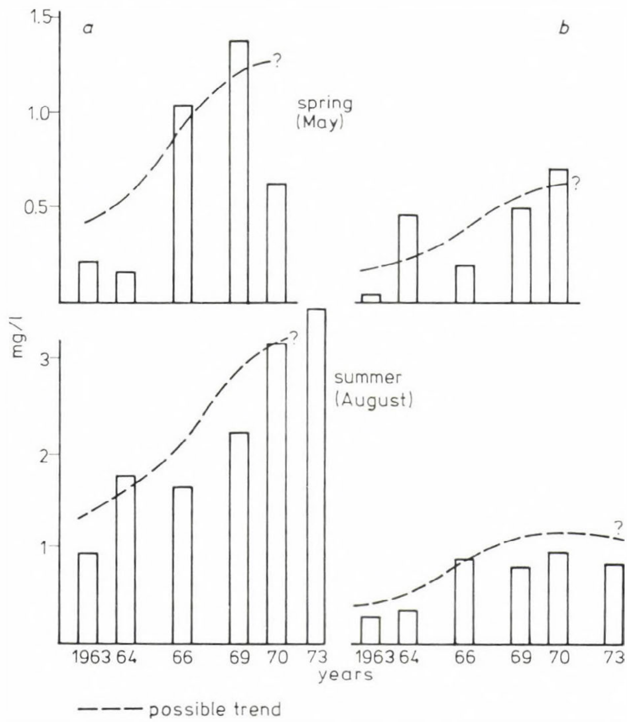

Fig. 5. Average biomass of filter-feeding crustaceans (*a*) and predatory Cyclopidae (*b*) in the epilimnion in two seasons of different years (Lake Mikołajskie)

* *Eudiaptomus graciloides* (Lilljeborg), *Diaphanosoma brachyurum* Lievin, *Daphnia cucullata* Sars., *D. longispina* O. F. Müller, *Bosmina coregoni thersites* Baird, *B. longirostris* (O. F. Müller), *Chydorus sphaericus* (O. F. Müller).

** *Mesocyclops leuckartii* (Claus), *M. oithonoides* (Sars).

A comparison of data from 1966, those from 1963 and 1964 showed an equal increase in the number of crustaceans in the other seasons (Spodniewska et al. 1973). The average size of adult individuals of all crustaceans also showed a significant rising tendency both in the spring and summer periods in the decade analysed (Fig. 6).

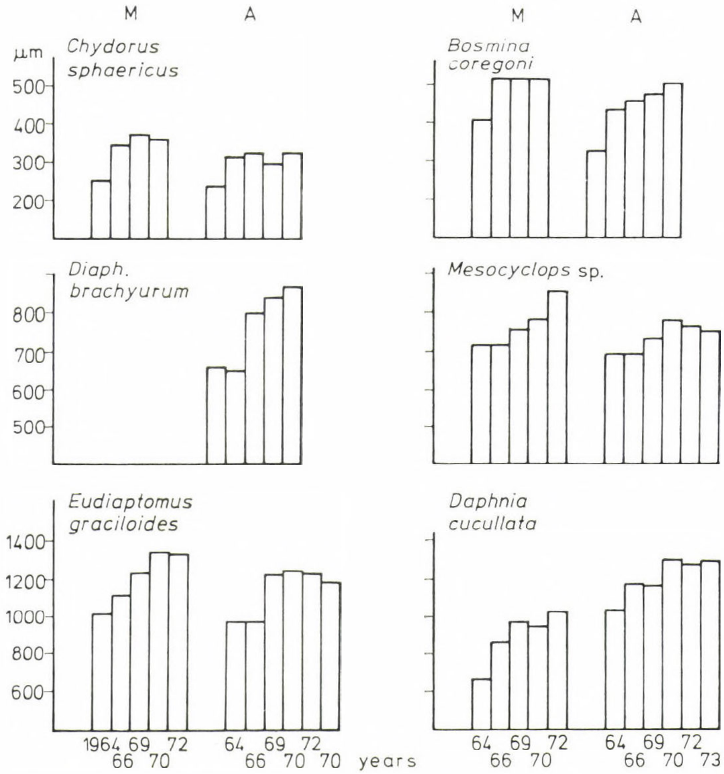

Fig. 6. Average size of adult individuals (μm) of plankton crustaceans in May (M) and August (A) in different years (Lake Mikołajskie)

No clear trend in changes of domination patterns of crustaceans could be observed in the study period (Fig. 2). Fluctuations in quantitative relations were rather remarkable from year to year, e.g. the very low abundance of *Bosmina longirostris* in 1964 (Fig. 7), but did not show a clear trend. Practically all species were included in the observed increase in biomass and individual size. Nevertheless, the disappearance was noted, probably accidentally (maybe due to the effect of different timing of biological cycles or by changing the sampling periods in successive years), a phenomenon rather difficult to explain. For example, *Bosmina coregoni* was abundant in the spring period of 1963 and 1964; in recent years, it has been replaced by *Daphnia cucullata; Daphnia longispina* was only abundant in summer 1963; in the following summer periods it occurred only sporadically, but in 1973 it appeared widely again as a significant contributor to the zooplankton biomass.

A significant increase of the biomass and the individual size of adult crustaceans were observed in the successive summer periods with a simultaneous reaction in the share of nannoplankton in food suspension. It seems

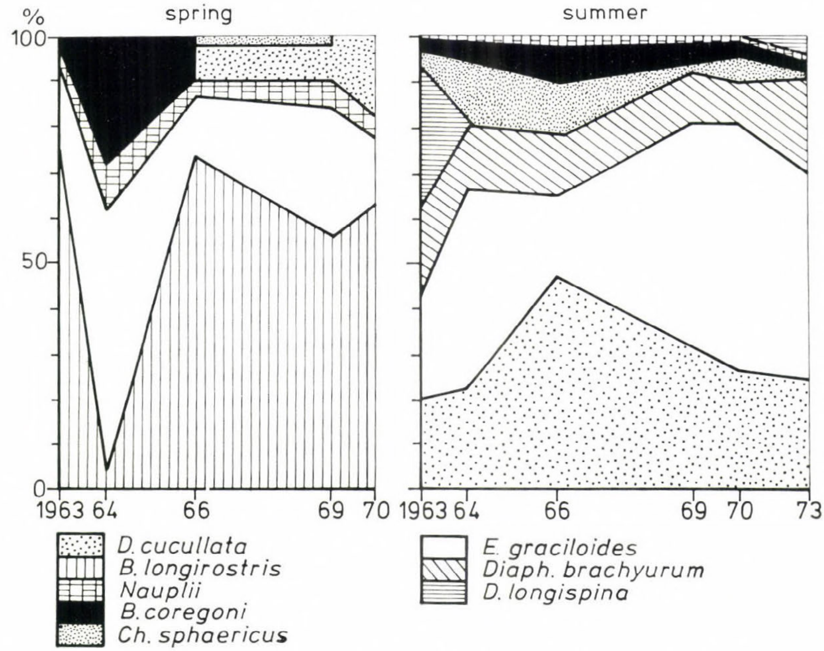

Fig. 7. Percentage composition of the biomass of filter-feeding crustaceans in two seasons of different years (Lake Mikołajskie)

that improved trophic conditions were rather due to an intensified production of detritus and bacteria, their minute particles dominating in food suspension. However, the permanently dominating species were *Eudiaptomus graciloides*, *Daphnia cucullata* and *Diaphanosoma brachyurum* and they are efficient filtrators feeding on particles of a wide size-range (up to 20 μm, Gliwicz 1969). No visible increase was observed in the number of zooplankton species like *Chydorus sphaericus* or *Bosmina coregoni*. These species are less efficient in filtration but feed preferably on smaller particles (up to 6–10 μm, Gliwicz 1969).

The observed increase of the biomass of filter-feeders seems to be one of the reasons for the observed increasing biomass of predatory Cyclopidae (Fig. 5). According to the experiments of Karabin (1978), these copepods can even catch cells of *Ceratium*. The increasing predation of these crustaceans could be the cause of the decreasing abundance of rotifers in the successive summer periods (Fig. 2). The preference of cyclopoids for small rotifers as food was documented by Karabin (1978).

The reason for the firm domination pattern of a filter-feeding crustacean community throughout the decade could be as follows.

According to Gliwicz and Prejs (1977), the rate of predation of planktivorous fish on zooplankton in Lake Mikołajskie is insignificant. These authors calculated the total consumption of bleak, smelt and vendace, and found that this constitutes a very small share in the total mortality of zooplankton biomass. Catches of the fishes mentioned above do not show any definite falling trend in Lake Mikołajskie, although fluctuating from year to year. This could account for both the increased biomass and the growth of specific size of crustaceans as a result of the reduced predation of these fishes.

Thus it appears that in Lake Mikołajskie the species composition of crustaceans is not at all affected by fish. Under these conditions, there are no factors suitable for eliminating selectively the larger zooplankton species like calanoids and daphnids. According to the so-called 'size-efficiency' hypothesis of Brooks and Dodson (1965) the lack of intensive fish predation is an important factor enabling the larger zooplankton species to successfully compete with smaller forms. According to these authors, this is due to the fact that larger zooplankton remove food particles from the water more efficiently (the filtering capacities being related to body size and filtering surface area) and can also catch the larger particles (up to 15–20 μm) not only the smaller ones (< 5 μm) for which all planktonic filter-feeders are in competition. The size-efficiency hypothesis is widespread among ecologists trying to explain the species composition of plankton-crustacean communities under conditions of different rates of fish predation. Recently, however, this hypothesis has been criticized (Hall et al. 1976).

Thus the permanent domination of calanoids and daphnids over small cladocerans (*Bosmina*, *Chydorus*) and rotifers in summer plankton in Lake Mikołajskie should mainly be induced by their higher filtering capacities. The latter could maintain the fecundity and growth of populations even if food suspension is composed mostly of bacteria and detritus, i.e. small-sized particles. Although these filter-feeders can retain bigger particles, they can similarly manage to remove the smaller ones (Gliwicz 1969). In other words, the permanent domination of these species in Lake Mikołajskie in the decade analysed was the result of the lack of effective fish predation, while the increasing biomass and specific size of crustaceans was mainly the result of a greater density of food suspension composed mostly of the products of algal decomposition.

On the basis of the size-efficiency hypothesis, the replacement of *Bosmina coregoni* (smaller species) by *Daphnia cucullata* (bigger species) in the spring period can also be explained (Fig. 7). Probably, *Daphnia cucullata* could manage better under conditions of increasing biomass of nannoplanktonic cells bigger than bacterial cells and detritus particles.

Gliwicz (1977) found that large-sized algae, especially filamentous blue-greens, can seriously affect the filtration mechanism of some filter-feeders and make the process of retaining the particles and feeding difficult. Not all species of filter-feeding crustaceans are equally sensitive to the damaging effect of filamentous algae. In species like *Daphnia longispina*, *D. cucullata* and *Bosmina coregoni*, the broad ventral crevice between the two halves of the carapax allows the filaments of algae and other large-celled algae (particularly damaging are those with sharp, hard spines, etc.) to enter the filtering chamber and then to block the process of retaining the particles on the setae. Thus, these species have an ineffective pre-selection mechanism. Other species like *Chydorus sphaericus*, *Diaphanosoma brachyurum*, *Bosmina longirostris* (probably?) having a narrow ventral crevice, can manage quite well in the presence of large-sized or filamentous algae. According to Gliwicz (1977), the same mechanism is supposed to be active during the process of lake eutrophication, i.e. an increase in the biomass of large 'non-consumable' phytoplankton, especially blue-green algae, causing a shift in the species composition by decreasing the abundance of such species as *Daphnia cucullata*, *Bosmina coregoni* and increasing the contribution of *Chydorus sphaericus*, *Diaphanosoma brachyurum*. Such trend was not detected in the analysed decade in Lake Mikołajskie, probably because the increase in *Ceratium hirundinella* was not as important as the parallel decrease of blue-greens.

INTERRELATIONS BETWEEN PHYTOPLANKTON AND ZOOPLANKTON IN LAKES WITH DIFFERENT PHYTOPLANKTON DENSITY

Data on the composition and biomass of phytoplankton and plankton-crustaceans from 13 Masurian Lakes including Lake Mikołajskie, collected in August 1973 (Table 1) (Spodniewska 1977, Węgleńska, unpublished data),

TABLE 1

Morphometric Data and Plankton Characteristics (August 1973, Epilimnion) for 13 Deep Lakes

No.	Lake	Surface (ha)	Depth (m)		Total Phytopl. biomass*		Nannopl. biomass*		Crustacean biomass (mg/l)	
			Maximum	Average	mg/l	Domination (above 40% of biomass)	mg/l	% in phytopl. biomass	Filter-feeders**	Predators+
1	Bełdany	780.0	31.0	10.0	8.7	dinofl.	1.2	13	3.57	0.25
2	Boczne	189.8	15.0	8.7	15.5	dinofl.	1.6	10	4.96	0.61
3	Guzianka Mała	42.0	14.0	2.6	2.6	nannopl.	2.1	79	2.35	0.12
4	Guzianka Wielka	72.0	29.4	6.5	12.1	nannopl. blue-greens	4.7	39	4.47	0.53
5	Jagodne	936.0	34.0	8.7	15.6	blue-greens	0.6	4	4.66	0.65
6	Mikołajskie	460.0	27.8	11.1	9.3	dinofl.	1.7	18	3.80	0.52
7	Nidzkie	1724.0	25.0	6.2	5.1	dinofl.	0.3	6	2.11	0.57
8	Niegocin	2498.8	40.0	10.0	7.1	diatoms	2.3	32	2.53	0.47
9	Ryńskie	620.0	47.0	13.6	34.2	dinofl.	1.9	6	5.14	0.95
10	Śniardwy	10 598.4	25.0	5.9	8.1	blue-greens	1.8	32	3.16	0.12
11	Tajty	251.2	34.0	7.6	5.5	dinofl.	1.3	23	1.60	0.39
12	Tałtowisko	323.4	35.0	14.0	6.3	dinofl.	0.8	13	2.37	0.51
13	Tałty	1162.0	37.5	13.6	8.5	dinofl.	1.3	15	3.02	0.60

* Data of Spodniewska (1977).
** *Eudiaptomus graciloides*, *E. gracilis*, *Daphnia cucullata*, *D longispina*, *Diaphanosoma brachyurum*, *Bosmina coregoni thersites*, *Chydorus sphaericus*.
\+ *Mesocyclops oithonoides*, *M. leuckartii*.

were used to analyse the interrelations between phytoplankton and zooplankton with view to the increasing biomass of algae. All analysed lakes have an average depth of up to 14 m, and are, because of summer stratification, comparable with Lake Mikołajskie.

Summer phytoplankton biomass varies in these lakes between 2.6 and 34.0 mg per l fresh weight and the biomass of filtering crustaceans varies from 1.6 to 5.14 mg per l fresh weight (Table 1). When taking the value of 8–10 mg per l of fresh algal biomass (in summer, surface water) as a limiting value between eutrophic and meso-oligotrophic lakes (according to Vollenweider 1971), it was found that eight of the analysed lakes were eutrophic (Table 1). In almost all the lakes the biomass of nannoplankton was 2 mg per l or less, and *Ceratium hirundinella* was found to be dominant in the algal biomass (Table 1).

There was no significant correlation ($r = 0.31$, $P = 0.05$) between the biomass of filter feeding crustaceans and the biomass of nannoplankton (Fig. 8), but rather a strong correlation ($r = 0.83$, $P = 0.05$) was obtained with the total algal biomass.

Similar results were obtained in Lake Mikołajskie as shown by the summarized data of successive spring and summer periods (Fig. 8): no significant correlation existed between the biomass of filter feeders and the nannoplankton ($r = 0.25$, $P = 0.05$) and a stronger correlation could be detected (but still on the level of significance) between filter-feeders and the total phytoplankton biomass ($r = 0.58$, $P = 0.05$).

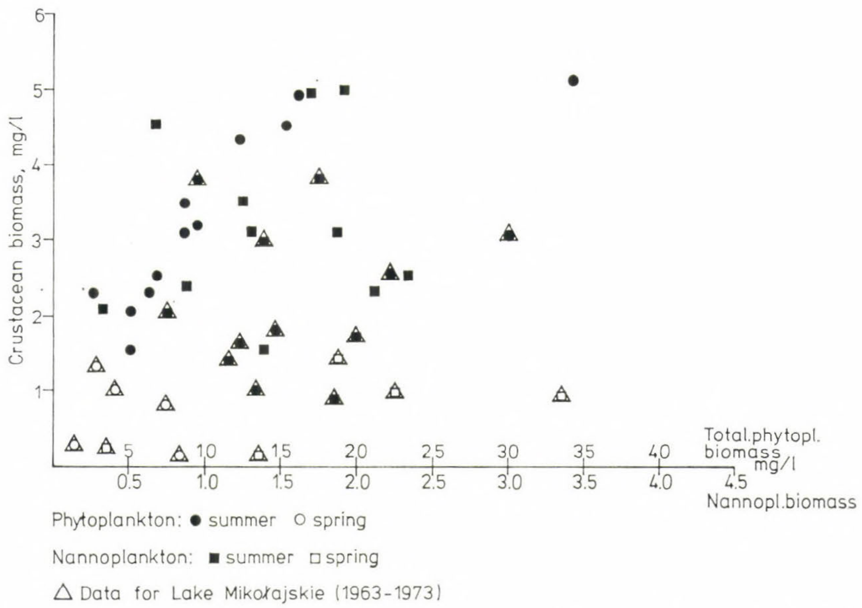

Fig. 8. Relation between biomass of filter-feeding crustaceans and total phytoplankton and nannoplankton biomasses in the epilimnion of 12 lakes in summer 1973 and of Lake Mikołajskie (in the summer and spring periods)

These correlations show that an increasing biomass of the total phytoplankton community, although *in vivo* not available for the zooplankton, is a better indicator of a changing trophic situation (because of an increase of the detrito-bacterial suspension) than the nannoplankton biomass on its own, which is a consumable but almost insignificant component of the phytoplankton community.

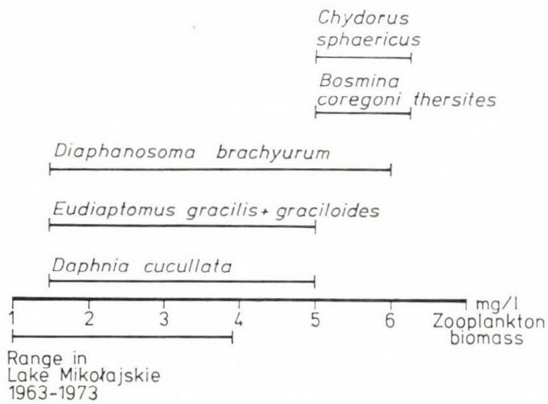

Fig. 9. Range of domination ($\geqslant 30$ per cent) of filter-feeding crustaceans in their summer biomass (based on data from 13 lakes)

Analysis of the dominance patterns within the crustacean communities shows (Fig. 9) that in the range of lower biomass values, up to 4–5 mg per l generally corresponding to lower biomass values of the phytoplankton (see Fig. 8), species as *Daphnia cucullata*, *Eudiaptomus gracilis* or *Eudiaptomus graciloides* and *Diaphanosoma brachyurum* have the greatest share. When biomass values are higher, above 5 mg per l, *Bosmina coregoni thersites* and *Chydorus sphaericus* join the species mentioned above. These two species are dominant in lakes with zooplankton biomass of about 6 mg per l and a phytoplankton biomass up to 35 mg per l. It should be mentioned that the coexistence of *Daphnia cucullata*, *Eudiaptomus graciloides* and *Diaphanosoma brachyurum* was also observed in Lake Mikołajskie in the period 1963–1973.

A rather strong correlation ($r = 0.70$; $P = 0.05$) could be noticed between the phytoplankton biomass and the biomass of predatory Cyclopidae (Fig. 10). An increase in the biomass of Cyclopidae seems to be related to the increase of both abundance and availability of prey organisms, i.e. small-sized zooplankton organisms. The trophic reaction to the increased abundance of dinoflagellates cannot be excluded, since according to Karabin (1978), Cyclopidae may even hunt for large cells of *Ceratium hirundinella*.

The long-term (Lake Mikołajskie) and comparative analysis of phyto- and zooplankton relationships shows that there are obvious correlations between lake fertility manifested by the increased phytoplankton biomass in summer and the abundance of primary consumers, i.e. the suspension-feeders. However, the tendencies of these two increasing factors are different and so with progressing eutrophication, the ratio of zooplankton biomass to phytoplankton biomass is clearly decreasing (Fig. 11). In other words, a lower zooplankton biomass is produced by the increasing biomass of phytoplankton which proves the decreasing efficiency of utilization of primary production by these consumers. This observation, based on the analysis of the ratio of zooplankton production to phytoplankton production, has been confirmed for various lakes investigated in different parts of the world in the course of the International Biological Programme (Hillbricht-Ilkowska 1977).

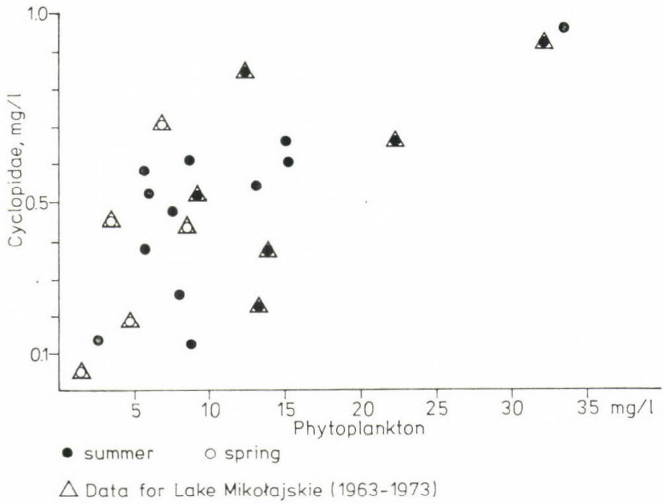

Fig. 10. Relation between biomass of predatory Cyclopidae and phytoplankton biomass in the epilimnion of 12 lakes (in summer 1973) and Lake Mikołajskie (summer and spring periods)

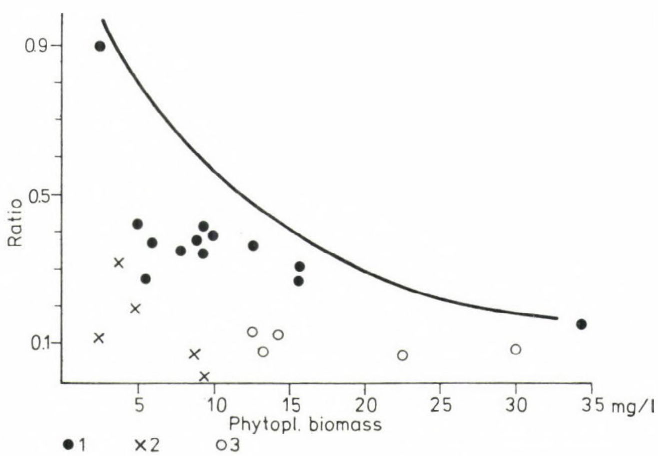

Fig. 11. Proportion of filtering crustaceans to phytoplankton biomass in 13 lakes in summer (1) and in Lake Mikołajskie in spring (2) and summer (3) periods of different years

SUMMARY

Analysis of the changes in phyto-zooplankton relationship during the decade 1963–1973 in Lake Mikołajskie and in other, stratified lakes with different phytoplankton biomass in summer, has suggested the following conclusions:

1. The increasing biomass of large-cell components of net phytoplankton during summer (in the lakes analysed almost exclusively *Ceratium hirundinella*) is a typical sign of eutrophication of deeper lakes. Although this involves algae 'non-consumable' *in vivo* by the suspension-feeders, it creates conditions favouring the development of filtering crustaceans (increase in their biomass and in the size of the individuals). The mechanism responsible for this phenomenon is probably the intensification of the production of small-sized food particles: detritus and bacteria, as products of decomposing algae.

2. In periods of very high summer biomass of net phytoplankton ($\geqslant 30$ mg per l) a shift within the dominating suspension-feeders is observed. Large filter-feeders such as *Eudiaptomus gracilis* or *Eudiaptomus graciloides* and *Daphnia cucullata* are replaced by small-sized *Chydorus sphaericus* and *Bosmina coregoni thersites*, and a typical summer species *Diaphanosoma brachyurum* occurs frequently at both high and low biomass of net phytoplankton. This trend seems to support the hypothesis of Gliwicz (1977) that species of the genus *Daphnia* and also *Bosmina coregoni* are sensitive to higher concentrations of large cells (especially blue-green algae) in their filtering chamber (wide ventral crevice between the halves of carapax not providing an effective pre-selection mechanism), while other forms, such as the genera *Diaphanosoma* and *Chydorus*, due to their ability of sufficient pre-selection (narrow ventral crevice), are less sensitive to these circumstances. This shift in the species composition and dominance was, however, not observed in Lake Mikołajskie during the period 1963–1973 when the algal biomass reached 30 mg per l in summer. Possibly, as also suggested by Gliwicz (1977), the occurrence of filamentous blue-green algae (even in smaller concentrations) have a greater influence on the zooplankton than the higher concentrations of dinoflagellates. The abundance of blue-green algae in Lake Mikołajskie diminished in the successive summer periods of the decade analysed.

3. It appears that the growth of food resource due to eutrophication affects the zooplankton more than any of the other possible factors, which are not connected with eutrophication either directly or indirectly, such as the changes in predation of planktivorous fishes (relying mainly on results of Lake Mikołajskie). The weak selective predation of fish probably accounts for the permanent domination of large zooplankton like *Eudiaptomus graciloides, Daphnia cucullata* in the summer periods of the decade analysed.

4. An increase of lake fertility due to eutrophication affects also the spring peak of phytoplankton development. An increasing abundance of small nannoplanktonic algae up to 30 μm has been observed. This, in turn, explains the increasing biomass of suspension feeders (crustaceans) and some of the raptatorial rotifers. This increase of zooplankton biomass is, however, less pronounced than in summer, probably as a result of the lower

rate of changes in the algal biomass in the successive spring seasons as compared to the summer periods.

5. An increasing abundance and the growing size of individuals of predatory Cyclopidae observed both in Lake Mikołajskie and in other lakes with different phytoplankton and zooplankton biomasses in summer, are most probably related to the greater abundance of suspension-feeders together with dinoflagellates, these species being the main prey organisms.

6. Induced eutrophication increases the abundance of primary producers followed by an increased abundance of consumers at different rates. This leads to a decreased efficiency of organic matter transfer in the pelagic food relations.

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THE POSSIBLE USE OF FISH, ESPECIALLY SILVER CARP—*HYPOPHTHALMICHTHYS MOLITRIX* (VAL.)—TO OVERCOME WATER BLOOMS IN TEMPERATE WATER BODIES

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Abstract

Silver carp reared without other fish in experimental enclosures in a shallow lake decreased the biomass of phyto- and zooplankton. However, combined activity of silver and common carp in fish ponds resulted in an incrased phytoplankton biomass. Eliminating macrophytoplankton and some zooplankton, silver carp should theoretically promote the development of nannophytoplankton. By removing large amounts of organic matter to bottom deposits, it should also counteract water blooming in stratified or very calm water bodies during summer stagnation.

INTRODUCTION

Considering the worldwide progress of eutrophication there is a great need for the restoration and recultivation of excessively eutrophic or saprotrophic (Olszewski 1971) water bodies. Several technical and chemical methods are available to decrease the trophic state, e.g. the removal of hypolimnetic waters rich in nutrients (Olszewski 1973) as well as of the surface layers of bottom deposits rich in precipitated phosphorus (Björk 1972), further aeration of the hypolimnion or exploitation of algae, etc. Some of these methods are expensive, while others can only be applied under special circumstances (Hine and Dyer 1974).

Besides, restoration of the ecosystem may also be achieved by modifying its structure and function by changing the abundance of some groups of organisms (fish, planktonic invertebrate predators, benthic organisms).

Several benthic species, mainly crustaceans, have been extensively introduced into water bodies, particularly in the U.S.S.R. (Wiśniewski 1976), but little is known about their effect on the function of the respective ecosystems. No attempts have probably been made to influence the abundance of other invertebrates in ecosystems as a whole, except for experiments with *Chaoborus* carried out in enclosures (Kajak and Rybak 1979, Smyly 1976). The greatest success can be achieved by manipulating fish. In view of the income obtained from fish yield, this is the cheapest way of influencing the function of ecosystems, and at the same time, the most promising way of rehabilitation of water bodies; however, its reliability should be discussed in detail.

MANAGING OF THE ECOSYSTEMS BY FISH

Influencing ecosystems by fish may include: (*i*) Exploitation of food organisms and changing their quantitative relations; (*ii*) excretions (of all kinds) changing the water chemistry and the decomposition processes, (*iii*) stirring of the mud and changing the abioseston concentration in the water thus modifying light-conditions and other physical and chemical parameters; (*iv*) removal of organic matter with the fish yield and fish faeces.

There are three main groups of papers reporting on the influence of fish on ecosystems:

1. Those describing changes in the composition and dominance structure of zooplankton, i.e. a decrease in the number of individuals of large-sized species and an increase of small-sized species caused by selective feeding of fish (review, Hall et al. 1976).

2. Those reporting on increased trophicity due to the impact of fish. In a rather dense stock of benthophagous fish (500 kg per ha, bream, roach, crucian carp) in enclosures at 2 m deep lake sites, typical features of increased trophicity have been observed: intensive phytoplankton blooms, increasing total phosphorus concentrations, elimination of large zooplankton organism, etc. The situation was reversed after moving fish from stocked to control variant (Andersson et al. 1977). In fish ponds (about 0.7 m deep) stocked with common carp and silver carp, the biomass of phytoplankton increased similarly (Januszko 1974, Opuszyński 1971). On the other hand, removal of benthophagous and planktophagous fish from lakes (Stenson et al. 1977) or from a reservoir (Hrbaček et al. 1977) resulted in more oligotrophic conditions. A possible explanation of the increasing trophicity due to fish activity is the increased dominance of small zooplankton members caused by the food selection of fish. The amounts of phosphorus and nitrogen released by small crustaceans are manyfold the amounts released by larger ones dominating at low-rate grazing activity (Peters 1975). An alternative explanation is that fish stimulate the decomposition processes by producing faeces and by stirring up the mud, etc. The importance of these phenomena has, however, not yet been clarified; Lamarra (1975) for example showed that mechanical stirring of mud (once a week only!) did not increase the nutrient concentration in the water. More studies are needed along these lines.

3. Studies reporting on a decrease in trophic status as a result of fish activity. Doubling of fish stock by the introduction of common carp (total fish stock about 100 kg per ha) into a shallow (maximal depth ± 3 m) eutrophic lake with abundant submerged macrophytes, has resulted in the decrease of phytoplankton biomass and production as well as in essential changes in zooplankton, i.e. in a shift towards the dominance of small crustaceans, and an increase in length and fertility of all crustaceans. The phytoplankton decline was most probably due to increased turbidity caused by the stirring up of the mud by the carp, as well as to the exploitation of large crustaceans due to more intensive fish feeding (Kajak and Zawisza 1973). All these occurred, however, at a relatively small phytoplankton biomass (5.0 mg per l in the control unit, with normal fish stock) and at a low rate of gross primary production (2000 Kcal m^{-2} $year^{-1}$).

In plastic, plankton-tight enclosures (6 m^2 surface area each) at 1.5 m depth in a eutrophic lake, silver carp (450–1350 kg per ha) decreased the phytoplankton as well as the zooplankton biomass several times (Fig. 1).

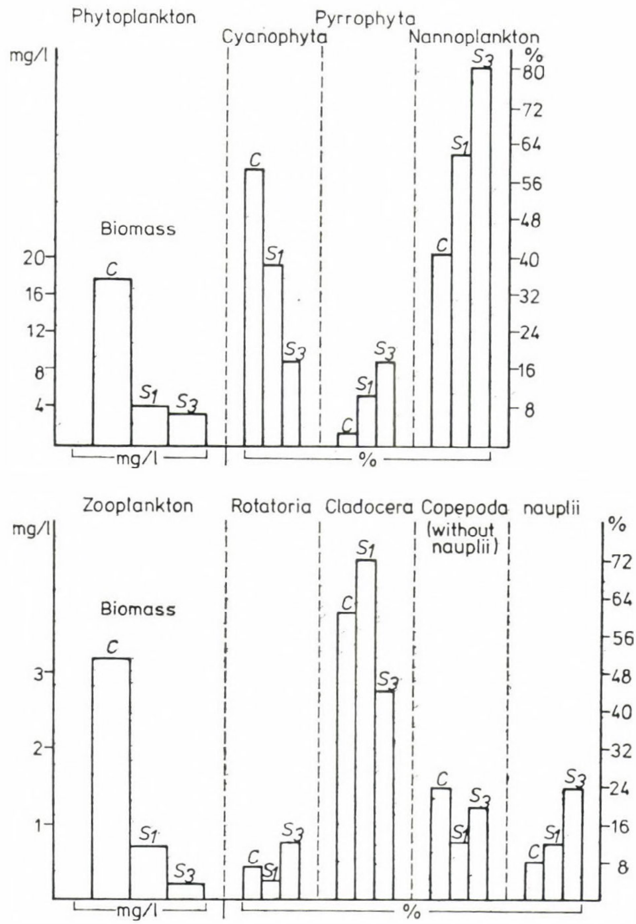

Fig. 1. Influence of silver carp on plankton biomass and share of particular groups; experiment in plankton-tight enclosures in the eutrophic Lake Warniak, at 1.5 m depth. C: control, without fish; S_1: with one specimen of silver carp per 1 m^3 (30 g per m^3); S_3: with three specimens of silver carp per 1 m^3 (90 g per m^3). Average for the period 5 July to 13 September 1973 (modified after Kajak et al. 1975)

The mean phytoplankton biomass was 16 mg per l in the control. In this relatively calm environment, with no other fish present, the nutrients were obviously removed with the faeces of silver carp and stabilized in the bottom deposits.

From this short review of the influence of fish on lake ecosystems, it seems that (*i*) fish increase the trophicity of shallow water bodies if there is a rather large fish stock composed at least partly of benthophagous species stirring up the bottom sediments and probably stimulating the decomposition and release of nutrients; (*ii*) some decrease in trophicity can be attributed to the activity of fish under specific circumstances.

Benthophagous fish may decrease the trophicity of a water body if their biomass is moderate or low. Obviously, under certain circumstances, the factors decreasing algal development (e.g. increased turbidity due to the stirring up of the mud by fish and the worsening of light conditions) may be more effective than the stimulating effect of nutrients released from the mud (Kajak and Zawisza 1973).

In the absence of other fish, particularly of benthophagous species, silver carp has been found to decrease the trophicity of the water body and to reduce algal biomass in calm or stratified water bodies where the faeces are removed from circulation.

Daily consumption rations of silver carp feeding mostly on phytoplankton are about 20 per cent of its biomass (Borutski 1973, Moskul 1977); this value is several times higher than that of zoophagous fish. As a result, also the production of faeces is several times higher. If the faeces remain in circulation, and particularly if they are stirred up by benthophagous fish together with the bottom deposits, an increased trophic level and phytoplankton biomass may result. If the faeces are removed from circulation, the water becomes poorer in nutrients and phytoplankton. The faeces of silver carp sink rather rapidly — 1 to 3 m sec^{-1} — (Barthelmes 1975) getting out of the epilimnion before significant decomposition and nutrient release can occur.

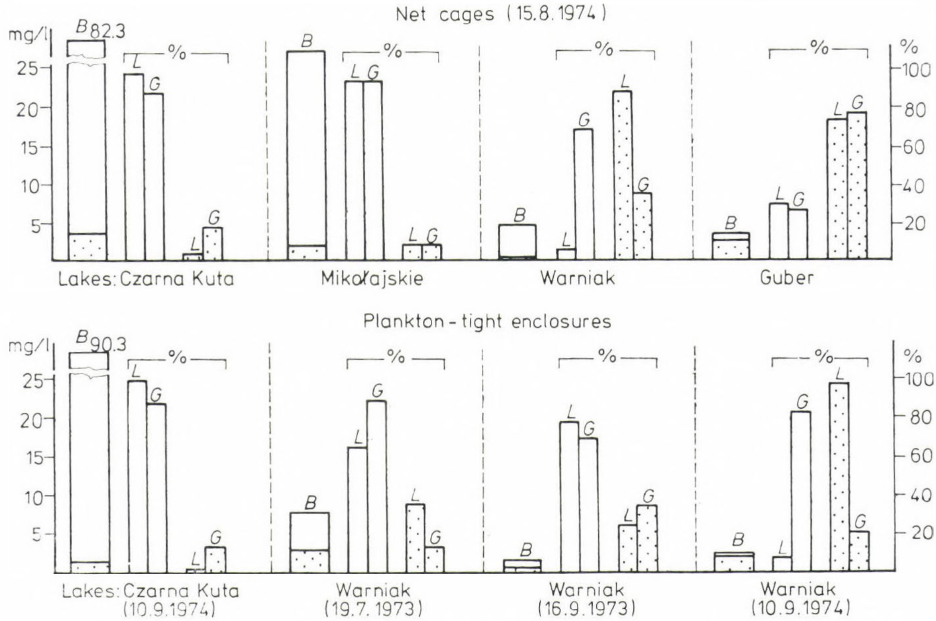

Fig. 2. Selective feeding of silver carp on phyto- and zooplankton in eutrophic lakes. Empty columns: phytoplankton; dotted columns: zooplankton. *B*: (thick columns) plankton biomass. Thin columns: per cent of phyto- and zooplankton in total plankton biomass in the lakes (*L*) as well as in the gut of fish (*G*) (modified after Kajak et al. 1977)

REMOVAL OF NUTRIENTS WITH FISH YIELD

It is usually assumed that in natural water bodies, with moderate fish yield, the amount of phosphorus removed with fish is small (Hine and Dyer 1974). However, in the eutrophic Lake Mikołajskie with a relatively high fish yield (about 100 kg per ha), in fact, a substantial part of the phosphorus load is removed in this way. The total allochthonous phosphorus load reaching the lake is 1583 kg yr^{-1}, deriving from sewage (1369 kg) and other sources (221 kg), i.e. surface runoff, erosion, etc. Although the 185 kg removed with the fish yield makes only 11.7 per cent of the total P load, without the sewage it is as much as 83.7 per cent (Kajak 1978). Theoretically, the yield of silver carp could be increased to surpass that of zoophagous fish as far as silver carp feeds mainly on plant material belonging to the first trophic level.

No high yield has so far been obtained with silver carp in the natural temperate water bodies, and studies on this subject are relatively few in number in Europe. High yields can be obtained in fish-ponds, with common carp and silver carp being reared together, and no other means of intensification being applied (Opuszyński 1978, Barthelmes 1976).

Taking an average of 0.4 per cent phosphorus in the fish (fresh weight), at a yield of 1000 kg per ha, a removal of 0.4 g phosphorus per m^2 can be expected. This is, according to Vollenweider (1968), several times more than the dangerous load for lakes of an average depth of 10 m. Thus such a fish yield may be helpful in decreasing trophicity.

FEEDING OF THE SILVER CARP

Although phytoplankton is usually the main component of the food of silver carp, both zooplankton and detritus are often dominant (Borutski 1973, Kajak et al. 1977, Lubeznov 1974, Opuszyński 1971, Sirenko et al. 1976, Vovk 1974). The percentage of phytoplankton and zooplankton in the gut contents in relation to their percentage in the water varies in the various environments (Fig. 2). The contribution of detritus is more important when the plankton is scarce. The growth of silver carp seems to be satisfactory while feeding on detritus (Omarov and Lazareva 1974). Rearing silver carp even in environments periodically or permanently poorer in plankton has yielded good results.

Of phytoplankton, blue-greens are often avoided (but eaten when very abundant and dominant) and diatoms are preferred (Kajak et al. 1977, Omarov and Lazareva 1974, Sirenko et al. 1976, Vovk 1974). However, the selection of particular groups and species of phytoplankton as well as the ratio of the main food components (phytoplankton, detritus, zooplankton) vary greatly, and it is difficult to survey the general patterns at the present state of our knowledge.

Feeding on bigger-sized phytoplankton organisms, silver carp does not compete with herbivorous zooplankton, which consume mainly smaller particles. On the contrary, by removing bigger phytoplankton organisms, silver carp can improve the conditions for small phytoplankton and stimulate its development.

It seems that the percentage of phyto- and zooplankton is usually similar in the gut contents of the silver carp and the environment (Kajak et al. 1977). The rate of turnover is higher for macrophytoplankton than for zooplankton. It can be concluded that the grazing effect of the silver carp is stronger on zooplankton than on phytoplankton. This creates an even more favourable situation for the small-sized phytoplankton since not only the numbers of its competitor (macrophytoplankton), but also those of its consumer (zooplankton) are diminished.

There are at present controversial opinions concerning the selective consumption of zooplankton by silver carp. Maltsman (1970) and Opuszyński (1978) have shown selective feeding of silver carp on bigger-sized forms. Grygierek (1973) found the opposite. Kajak et al. (1975) found only a weak influence of silver carp on the composition of zooplankton (e.g. the share of nauplii increased clearly), while there was a 16-fold decrease in the biomass of zooplankton (Fig. 1). The zooplankton almost disappeared from the fish-ponds with high silver carp densities (Opuszyński 1978). It seems that selective feeding on zooplankton of silver carp is very changeable, depending on the circumstances.

The abundance and composition of phytoplankton due to the selective feeding of silver carp will obviously depend on the intensity and selectivity of the exploitation of phytoplankton and zooplankton, on the abundance, quantitative relations within and between these groups, and on environmental conditions.

Filtratory zooplankton can be very important in the removal of phytoplankton. In highly eutrophic environments the biomass of the filtratory zooplankton as well as that of the silver carp can be very high. The food rations of filtratory zooplankton are about 100 to 200 per cent of its own biomass (Hillbricht-Ilkowska 1976), up to 10 times higher than the food rations of silver carp. However, it must be stressed that silver carp and zooplankton crustaceans feed on particles different in size (Table 1). This means

TABLE 1

Biomass and Consumption of the Most Important Planktonic Consumers and Their Pressure on Plankton (Situation in the Epilimnion of 5 m Depth)

Consumer	24h consumption in per cent of consumer biomass	Biomass of filtrators		Consumed seston $g\ m^{-3}\ 24\ h^{-1}$	% of seston exploited during 24 h, at seston biomass[d]		Size of particles consumed, μm
		$g\ m^{-2}$	$g\ m^{-3}$		$30\ g\ m^{-3}$	$100\ g\ m^{-3}$	
Filtratory		30	6.0[a]	9	30	9	$\leqslant 30$
crustaceans	150	150	30	45	150	45	
Filtratory		2[c]	0.4[a]	1	3	1	$\leqslant 5$–7
rotifers	250	100	20	50	167	50	
Silver carp	20	3	0.6[b]	0.12	0.4	0.13	
		100	20.0	4.0	13.3	4	$\geqslant 20$
		500	100.0	20.0	66.7	20	

a Average and close to maximum biomass in eutrophic water, respectively.
b 30, 1000, 5000 kg ha^{-1}, respectively.
c Very high abundance of rotifers occurs rather alternatively with that of crustaceans.
d Although usually phytoplankton dominates in the gut of silver carp, it never constitutes 100 per cent of gut contents, sometimes being only a small part thereof. Thus the pressure of silver carp on phytoplankton is smaller than on the total seston.

that the removal of a certain amount of zooplankton by silver carp results in a very significant decrease of the impact on small-sized phytoplankton, stimulating its development which is advantageous as far as water purity is concerned. But owing to very complicated relationships within planktonic communities, including feeding, competition, excretion to the environment, etc., at present it is impossible to predict the consequences of the removal of both phytoplankton and zooplankton by silver carp. Moreover changes in the bottom deposits will induce changes in the plankton (Fig. 3). The indirect effects of silver carp activity are doubtless more important than the direct exploitation of plankton by this fish species (Kajak 1977, Opuszyński 1978).

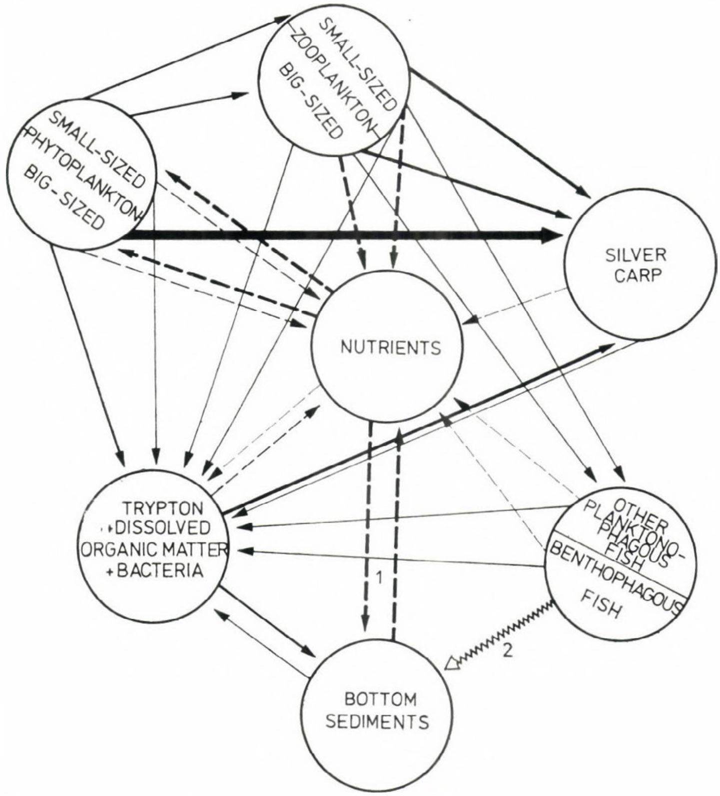

Fig. 3. More important connections of silver carp with other components of the ecosystem. Continuous lines: trophic relations and circulation of organic matter. Broken lines: circulation of nutrients. Zig-zag lines: non-trophic relationships (transformation of environment influencing decomposition and release of nutrients). 1 = Mostly to the bottom in stratified lakes during stagnation periods; 2 = most important in shallow water bodies

SUMMARY

Silver carp grows well in both the fish-ponds and lakes of Central Europe (Barthelmes 1976, Bryliński et al. 1976, Opuszyński 1971). It does not compete with zooplankton, feeding on particles larger than those available for zooplankton. There can be only some overlapping with copepods and big cladocerans. It removes some zooplankton, thus decreasing the grazing intensity affecting small-sized phytoplankton (nannoplankton). Zooplankton is more strongly affected by fish than is phytoplankton. The removal of macrophytoplankton (a competitor of nannoplankton) and zooplankton (a consumer of nannophytoplankton) by silver carp may achieve cumulative effects, stimulating the development of nannoplankton, provided that not too much nutrient is removed from the system. By a non-selective exploitation of plankton-crustaceans, as far as the size of food organisms is concerned, silver carp (at a moderate density) should change neither the composition nor the dominance of zooplankton, nor the intensity of the circulation of nutrients, provided the zooplankton biomass does not decrease significantly.

In stratified water bodies substantial amounts of organic matter are removed (much more than in the case of zoophagous fish) from circulation with the sedimenting faeces of silver carp, at least in the period of stagnation.

In shallow polymictic water bodies silver carp seems to increase rather than to decrease the biomass of phytoplankton. However, at the same time, it changes the dominance structure of phytoplankton; blue-greens disappear or become very scarce, and other groups (mostly smaller forms) become dominant (Januszko 1974, Kajak et al. 1975), which is desirable as regards water purity.

Silver carp can most probably survive the periodic scarcities of phytoplankton, feeding on detritus during that time.

Concerning the feeding of silver carp, one of the most uncertain points is its food selection. It seems rather changeable, depending on many factors like the quality, composition and distribution of seston, etc. It cannot be excluded that in some situations silver carp avoids feeding on the most noxious algae. On the other hand, it is possible that, if its population is dense enough, it could also stop the development of water blooms at their early stages.

It is quite possible that due to the cumulative effect of the removal of phosphorus from the circulation (with faeces and with the fish yield), a silver carp stock may keep the water clean. Since many complex mechanisms and interrelations are involved in the processes determining phytoplankton development and water purity, still more work is needed for solving these problems with special regard to stratified water bodies (e.g. selective feeding, indirect influence on phytoplankton by eliminating zooplankton, sedimentation, phosphorus release by planktonic communities, etc.).

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SOME REMARKS ON THE RECENT EVOLUTION OF THE DEEP ITALIAN SUBALPINE LAKES*

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Abstract

The largest and deepest lakes belonging to the subalpine Italian district (Lakes Garda, Iseo, Como, Lugano and Maggiore) are compared as to their physical, chemical and biological features, considering the following aspects: (*i*) geochemical characteristics of the drainage basin; (*ii*) morphometric characteristics of the lakes; (*iii*) water residence time and overturn of each lake. An interpretation of their actual trophic state is made in the light of the modifications they have undergone during the last five years (1973–1977), including comparisons, when possible, with historical data.

INTRODUCTION

The major lake district in Italy lies to the south of the Alps. Here are located our deepest water bodies: they are, from east to west, Lakes Garda, Iseo, Como, Lugano and Maggiore.

These lakes exceed a total volume of 120 km^3, all within the River Po catchment area, and contribute water for about one-third of its discharge. The fact that the plain between the River Po and the subalpine lakes is the most densely populated, industrialized and agriculturally highly productive area in Italy gives cause for concern.

It is, therefore, extremely important that limnologists monitor the quality of water the level of eutrophication or other kinds of pollution in each of them. Knowledge of the climatic features, of the drainage basin — both from a geochemical and hydrological point of view — of human pressures, of the physical, chemical and biological aspects of each lake is necessary. Comparison between present and historical data should reveal the trend of evolution which they are undergoing.

This kind of research is not only very important for the improvement of our understanding of limnological processes, but also for urging politicians into action for the conservation or restoration of the lakes.

An introduction to the case-histories of these lakes is summarized in Fig. 1 (a map indicating location); in Fig. 2 (the hypsographic curves of the drainage basins) and in Table 1 (main characteristics of identification).

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TABLE 1

Main Identification Characteristics of the Subalpine Lakes

	L. GARDA	L. ISEO	L. COMO	L. LUGANO	L. LUGANO (NE)	L. MAGGIORE
Mean watershed altitude, m (a.s.l.)	966	1429	1569	786	—	1283
Mean lake level, m (a.s.l.)	65	186	198	271	271	194
Lake area (Ao), km²	370.0	61.8	145.9	48.9	27.9	212.5
Watershed area (A), km²	2350	1842	4572	615	307	6599
A/Ao	6.4	29.4	31.2	12.7	11.0	31.2
Maximum depth, m	346	251	410	288	288	370
Mean depth, m	136	123	153	134	160	177.5
Average outflow discharge, $m^3 sec^{-1}$	58.5	58.7	158.0	24.5	12.2	288.2
Lake volume, km³	49.0	7.6	22.5	6.6	4.4	37.5
Water residence time, years	26.6	4.1	4.5	8.5	11.4	4.1
Volume below 200 m depth, km³	8.3	0.7	4.0 2.9*	0.6	0.6	8.7
Area at 200 m depth, km²	98.20	17.67	52.62 23*	11.68	11.68	90.70

* Lake Como sub-basin figures.

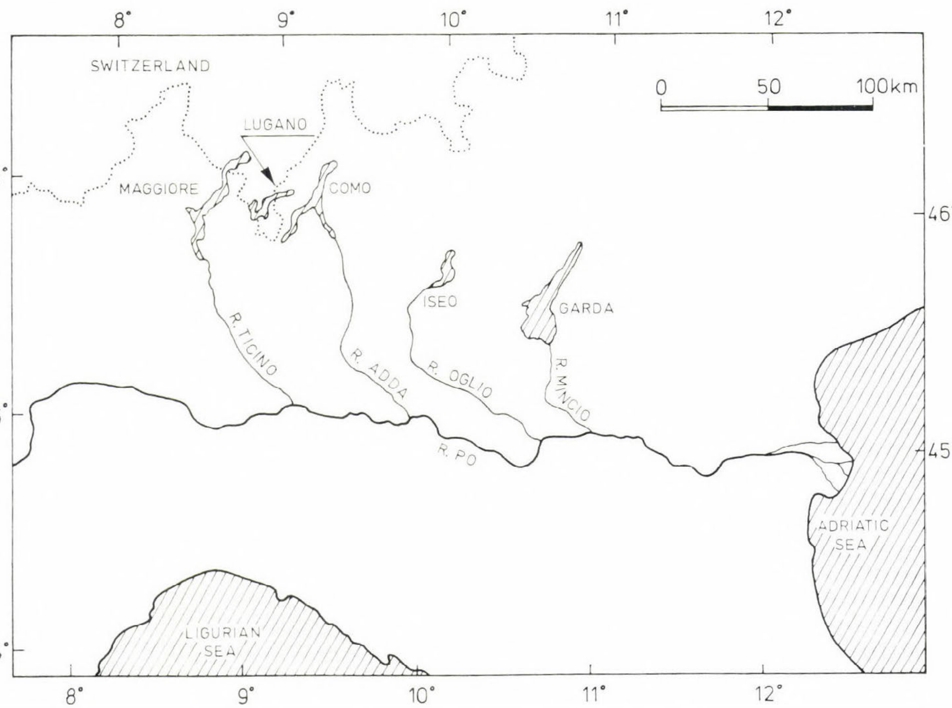

Fig. 1. Location of Italian subalpine lakes

The basic chemical characteristics, as well as the ionic spectra of lakes principally depend on the geochemical structure of their drainage basins.

The watersheds of Lakes Garda and Iseo are mainly characterized by sedimentary rocks, crystalline formations being present only in the most northern sections. The drainage basin of Lake Como shows a dominance, in the northern part, of igneous rocks, which outcrop also in the northern and western parts of the Lake Maggiore watershed. In both lakes the southern region is dominated by calcareous rocks.

The composition of Lake Lugano drainage basin, which is a part of the Lake Maggiore watershed, is more complicated: while limestones and dolomites prevail, outcrops of red quarziferous porphyries and metamorphic rocks also occur.

The ionic balance, conductivity and mean pH values are given in Table 2. Bicarbonates increase in the following order: Maggiore, Como, Iseo, Garda and Lugano, which is consistent with the structure of their watersheds. Maximum conductivity values were found in Lake Iseo where sulphate

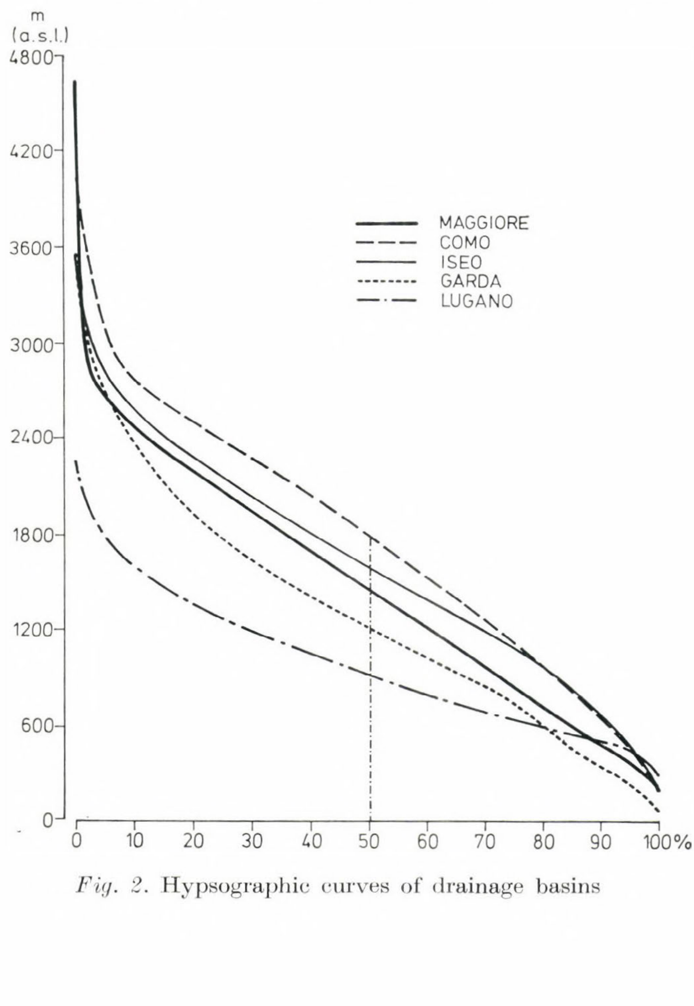

Fig. 2. Hypsographic curves of drainage basins

TABLE 2

pH, Conductivity (18 °C) and Ionic Balance

	L. MAGGIORE	L. LUGANO (NE)	L. COMO	L. ISEO	L. GARDA
pH	7.32	7.71	7.57	7.89	8.08
Cond., μS cm^{-1}	137	221	161	255	200
HCO_3, mEq l^{-1}	0.784	2.429	1.241	1.882	2.110
SO_4, mEq l^{-1}	0.598	0.262	0.518	0.978	0.227
Cl, mEq l^{-1}	0.065	0.089	0.061	0.089	0.134
NO_3, mEq l^{-1}	0.055	0.017	0.052	0.048	0.024
Σ Anion	1.502	2.797	1.872	2.997	2.495
Ca, mEq l^{-1}	1.104	1.860	1.312	2.238	1.609
Mg, mEq l^{-1}	0.315	0.776	0.469	0.701	0.696
Na, mEq l^{-1}	0.093	0.103	0.106	0.094	0.131
K, mEq l^{-1}	0.036	0.034	0.030	0.031	0.025
Σ Cation	1.548	2.784*	1.917	3.064	2.461

* Included ammonia salts (0.011 mEq l^{-1}).

attains the highest concentration because of the presence of gypsum rocks in its drainage basin.

The pH values, calculated as means for the last 5 years (1973–1977), clearly show a close relation with bicarbonate concentrations (total alkalinity).

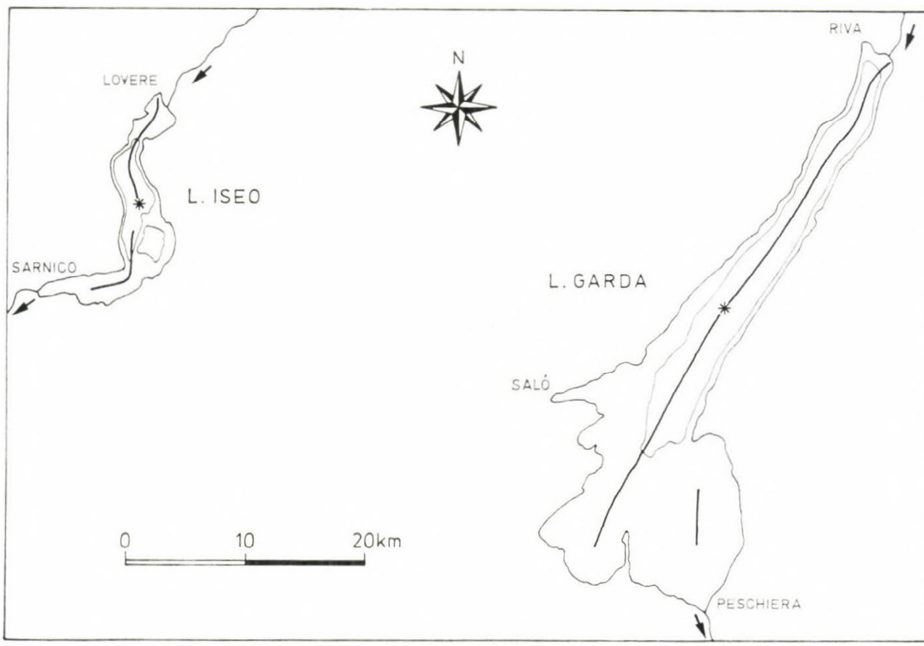

Fig. 3. Water sampling stations (maximum depth) and benthos sampling profiles

TABLE 3

Temperature and Main Chemical Variables of Lake Garda

Depth	25/3 1973	26/3 1974	25/3 1975	17/3 1976	21/4 1977	25/3 1973	26/3 1974	25/3 1975	17/3 1976	21/4 1977
	Temperature (°C)					pH				
0	8.5	9.0	8.3	8.2	8.9	8.3	8.3	8.2	8.0	8.2
10	8.1	8.8	8.3	8.2	—	8.2	8.2	8.2	8.1	—
30	8.0	8.5	8.3	8.1	8.6	8.2	8.2	8.2	8.1	8.2
50	8.0	8.2	8.3	8.1	8.4	8.1	8.1	8.2	8.1	8.2
100	7.9	8.1	8.3	8.1	8.3	8.2	7.9	8.1	8.1	8.2
150	7.8	8.0	8.1	8.1	8.3	8.2	8.1	8.0	8.0	8.1
200	7.8	—	8.0	8.1	8.2	8.1	—	8.0	8.0	8.1
250	7.7	7.9	7.9	8.1	—	8.0	8.2	8.0	8.0	—
300	7.6	—	7.9	—	8.1	8.0	—	7.9	—	8.0
340	7.6	7.8	7.9	8.1	8.1	8.0	8.0	7.9	8.0	8.1
	Oxygen (mg l^{-1})					Oxygen (% sat.)				
0	11.6	11.4	11.4	10.6	11.2	103	103	101	94	101
10	11.5	11.5	11.3	10.6	—	101	103	100	94	—
30	11.2	11.4	11.2	10.3	10.8	99	101	100	91	96
50	11.2	10.8	11.1	10.2	10.5	99	96	99	90	93
100	11.0	10.7	10.6	10.1	10.4	96	95	94	89	92
150	10.6	10.1	9.5	9.9	10.4	93	89	84	87	92
200	10.5	—	9.0	9.4	9.6	92	—	79	83	85
250	9.3	9.1	8.6	8.4	—	81	80	75	74	—
300	8.9	—	8.2	—	9.1	78	—	72	—	80
340	8.7	8.8	8.1	7.9	9.0	76	77	71	70	80
	Nitrate (μg N l^{-1})					Silicate (mg Si l^{-1})				
0	310	300	270	320	330	0.7	0.3	0.4	0.9	0.7
10	290	290	270	300	—	0.7	0.4	0.4	0.9	—
30	320	300	290	320	330	0.7	0.5	0.2	0.9	0.7
50	290	330	290	310	350	0.7	0.5	0.3	0.9	0.7
100	320	340	300	320	340	0.7	0.6	0.5	1.0	0.8
150	320	340	230	330	360	0.8	0.5	0.7	1.0	0.9
200	340	—	290	330	340	0.8	—	0.8	1.1	0.9
250	340	330	320	320	—	1.1	0.8	0.6	1.5	—
300	370	—	350	—	380	1.3	—	0.9	—	1.0
340	370	370	360	350	360	1.5	0.9	0.9	1.6	1.1
	Reactive phosphorus (μg P l^{-1})					Total phosphorus (μg P l^{-1})				
0	6	0	0	6	0		0	2	7	5
10	3	0	4	1	—		0	4	8	—
30	5	0	1	2	1		0	3	7	7
50	2	0	5	4	0		0	5	4	4
100	6	3	6	2	0		3	6	7	2
150	6	0	7	3	1		1	8	5	5
200	3	—	7	6	1		—	8	13	9
250	11	3	12	10	—		3	12	15	—
300	12	—	14	—	8		—	15	—	22
340	24	12	6	10	3		18	17	17	6

The lower pH of Lake Lugano is an exception, attributed to a highly productive and practically meromictic state, resulting in increased CO_2 in the hypolimnion.

Lake Garda is formed by two distinct basins: the deeper one, a typical glacial valley ($z_{max} = 350$ m), is NNE-SSW oriented, while the second, shallower basin ($z_{max} = 81$ m) broadens the southern part of the lake to the SE (Fig. 3).

Water samples were taken at the maximum depth and benthos was collected along a bottom profile as indicated in Fig. 3. The results of chemical analyses are given in Table 3.

Lake Garda presents high bottom water temperature, our measurements ranging between 7.6 and 8.1 °C.

This may be due to the climatic and morphometric features of this lake: in fact, the highest mean annual air temperature at the lake level (Table 4),

TABLE 4

Average Air Temperature and Annual Thermal Range

	J	F	M	A	M	J	J	A	S	O	N	D	Year	Δt
L. Maggiore	2.8	4.1	8.1	12.2	16.3	20.4	22.8	22.1	19.2	13.3	8.0	3.8	12.7	20.0
L. Lugano	1.9	3.1	7.1	11.1	15.2	19.1	21.9	21.1	18.3	12.2	7.5	3.3	11.8	20.0
L. Como	3.4	4.7	8.6	12.4	15.9	20.6	23.2	22.2	19.6	13.7	8.6	4.4	13.1	19.8
L. Iseo	3.9	4.3	6.4	9.2	12.0	17.6	20.2	22.9	20.4	14.3	9.2	5.2	12.1	19.0
L. Garda	3.2	4.1	8.0	12.8	16.8	21.2	23.7	23.1	19.7	14.1	9.1	4.5	13.4	20.5

the lake altitude (65 m a.s.l.), the mean altitude of the watershed (Fig. 2), the very long water renewal time and the relatively little mean depth (Table 1) are to be considered favourable factors.

Table 3 shows the good chemical conditions of the lake: reactive and total phosphorus rarely exceed 10 and 15 μg P l^{-1}, respectively, and nitrates fluctuate around 340 μg N l^{-1}. The lake has a high hypolimnic O_2 content; a moderate oxygen decrease occurs only in the deepest waters.

The vertical oxygen distribution in early spring provides a suitable test for the vertical extension of water circulation.

Consequently, we believe that the last complete overturns took place in 1971 (Casellato and Duzzin 1974) and 1977 (Table 3). As will be shown later, not all of the deep lakes were characterized by total circulation in 1977; it must be remembered, on the other hand, that Lake Garda has rather windy conditions.

For instance, in 1971, for which wind data are available for two Garda stations (Riva del Garda, in the north; Sirmione, in the south), total windrun was 31 940 and 27 350 km yr^{-1}, respectively, while only 17 850 km yr^{-1} was recorded at the meteorological station of Pallanza, Lake Maggiore. As a consequence, thermal stratification is less stable when compared with the other large subalpine lakes. This was also noticed during a previous investigation of Lake Garda by Merlo and Mozzi (1963).

The very good oxygenation of the profundal waters of Lake Garda depends also on its low productivity. Phytoplankton production, first measured in 1971–1972 (Gerletti 1974) allows the lake to be classified as oligo-

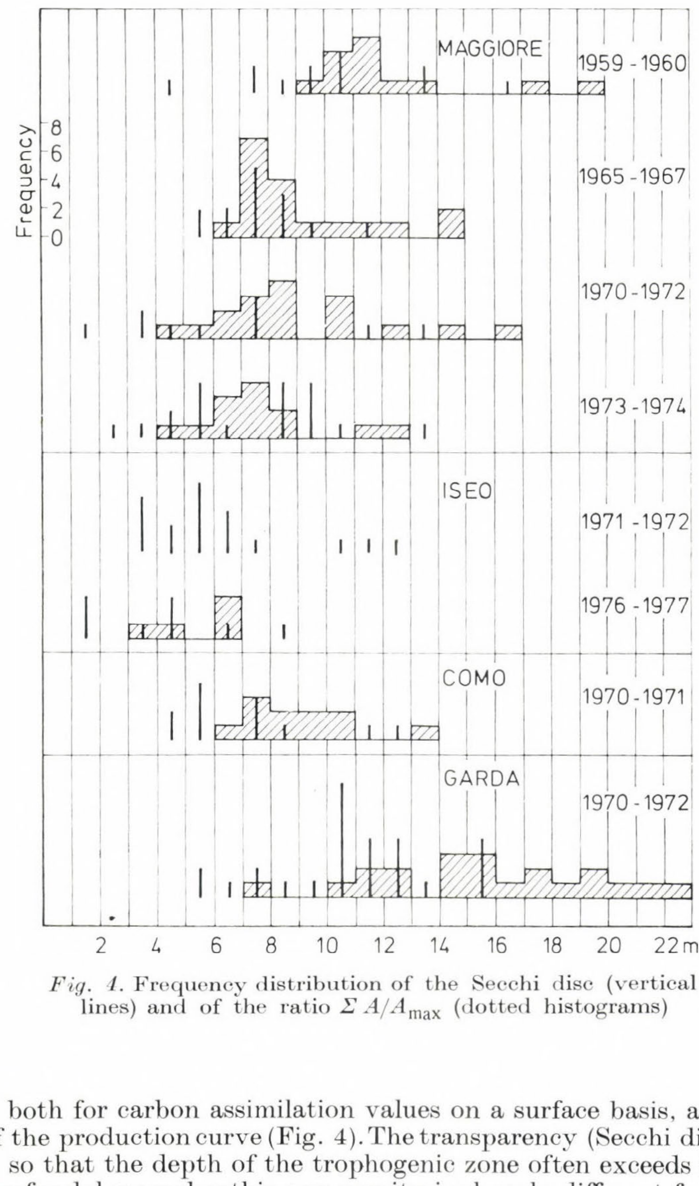

Fig. 4. Frequency distribution of the Secchi disc (vertical lines) and of the ratio $\Sigma A/A_{max}$ (dotted histograms)

trophic, both for carbon assimilation values on a surface basis, and for the shape of the production curve (Fig. 4). The transparency (Secchi disc) is high (Fig. 4), so that the depth of the trophogenic zone often exceeds 30 m.

The profundal macrobenthic community is sharply different from that of the other lakes. The presence of three amphipod species, the rather low densities, the high biotic diversity values, which moderately decrease with depth, are strongly consistent with the above-mentioned parameters in confirming the good general conditions of the lake (Bonomi 1974).

In general, the chemical situation, the phyto- and zooplankton communities are essentially the same as previously noted (Marchesoni 1952, Merlo and Mozzi 1963, Tiso 1962, Tonolli et al. 1975, Casellato and Duzzin 1974, Gerletti 1974).

During the last five years, however, the lake has been colonized by bivalve *Dreissena polymorpha* (Giusti and Oppi 1973, Bianchi et al. 1974) which, as usual, obstructs water pipes.

Lake Iseo

Lake Iseo (Fig. 3) attains a maximum depth of 250 m; the mean depth is 123 m and the lake water renewal time (4.1 years) is much shorter than that of Lake Garda (Table 1).

As in the other lakes, with the exception of Lake Lugano, the deep water temperature increased from 1973 to 1977 (6.0 to 6.5 °C, Table 5). In 1967

TABLE 5

Temperature and Main Chemical Variables of Lake Iseo

Depth	6/4 1973	27/3 1974	26/3 1975	16/3 1976	20/4 1977	6/4 1973	27/3 1974	26/3 1975	16/3 1976	20/4 1977
	Temperature (°C)					pH				
0	7.0	8.7	7.2	6.3	8.9	8.1	8.4	8.0	8.1	8.0
10	6.5	7.9	6.6	6.3	—	8.0	8.2	8.2	8.1	—
20	—	—	6.6	6.2	7.8	—	—	8.1	8.0	8.0
30	6.2	6.2	6.4	6.3	—	7.8	8.0	8.0	8.0	—
50	6.1	6.2	6.3	—	6.7	7.9	7.9	7.9	—	7.9
75	—	—	—	6.3	6.5	—	—	—	7.9	7.9
100	6.0	6.1	6.3	6.3	6.5	7.9	7.9	7.8	7.9	7.9
150	6.0	6.1	6.3	6.3	6.5	7.9	7.8	7.8	7.9	8.0
200	6.0	6.1	6.3	6.3	6.5	7.8	7.8	7.8	7.9	7.8
245	6.0	6.1	6.3	6.3	6.5	7.8	7.8	7.7	7.8	8.0
	Oxygen (mg l^{-1})					Oxygen (% sat.)				
0	11.2	12.8	11.8	11.1	11.9	98	116	104	96	109
10	11.0	11.8	12.0	11.0	—	97	105	103	95	—
20	—	—	11.4	10.7	11.6	—	—	99	92	103
30	10.3	10.1	10.2	10.1	—	88	87	88	87	—
50	9.7	9.4	9.8	—	8.9	83	81	84	—	77
75	—	—	—	7.4	7.6	—	—	—	64	66
100	8.8	7.0	7.1	5.0	6.7	75	60	61	43	58
150	7.5	6.2	5.7	4.4	5.3	64	53	49	38	46
200	6.9	5.8	5.4	3.8	3.9	59	50	47	33	34
245	6.1	4.9	5.2	3.1	3.9	52	42	45	27	34
	Nitrate (μg N l^{-1})					Silicate (mg Si l^{-1})				
0	630	660	580	580	610	1.4	0.7	0.4	0.8	1.1
10	630	610	610	600	—	1.5	0.8	0.4	0.8	—
20	—	—	580	590	630	—	—	0.3	0.9	1.1
30	660	670	560	570	—	1.6	1.1	0.8	1.0	—
50	660	670	560	—	740	1.6	1.1	0.7	—	1.4
75	—	—	—	680	740	—	—	—	1.8	1.6
100	700	680	590	700	750	1.7	1.0	1.1	2.3	1.6
150	760	660	550	690	780	1.9	1.4	1.0	2.4	1.7
200	700	660	580	730	750	1.9	1.3	1.3	2.5	1.9
245	700	670	570	730	760	2.0	1.5	1.3	2.6	2.2

TABLE 5 *(cont'd)*

Depth	6/4 1973	27/3 1974	26/3 1975	16/3 1976	20/4 1977	6/4 1973	27/3 1974	26/3 1975	16/3 1976	20/4 1977
	Reactive phosphorus (μg P l^{-1})					Total phosphorus (μg P l^{-1})				
0	13	0	4	3	8	26	6	5	10	14
10	14	9	4	2	—	31	9	7	11	—
20	—	—	3	2	6	—	—	3	14	12
30	11	13	8	4	—	22	19	9	10	—
50	19	2	11	—	19	26	7	11	—	23
75	—	—	—	20	25	—	—	—	22	28
100	18	15	28	41	30	30	19	29	43	37
150	24	30	45	48	37	32	38	45	50	40
200	28	35	45	51	41	33	41	46	55	44
245	28	26	48	63	50	37	30	48	73	53

Bonomi and Gerletti (1967) measured a bottom temperature of 5.75 °C. The mean oxygen concentration below 200 m depth gradually dropped from 6.46 mg l^{-1} in 1973 to 3.42 mg l^{-1} in 1976, and it rose again to 3.9 mg l^{-1} in April 1977 (Table 6). This moderate re-oxygenation of deep waters was the result of the exceptional winter cold water inflows from the drainage basin into the lake. Data obtained by the Consorzio dell'Oglio (local water-irrigation authority) show that the River Oglio winter discharge in 1977 was double that of the reference period (1933–1976).

During the last five years, the mean nitrate content has shown no definite trend, ranging from 0.57 to 0.74 mg N l^{-1}. Reactive and total phosphorus has shown a worrying rate of increase: 4.7 and 3.2 μg P l^{-1} yr^{-1} (for the whole water column) up to 1976.

Data on water chemistry before our investigations are rather scanty: Vollenweider (1965) reported on the ionic balance, while Bonomi and Gerletti (1967) gave the first general limnological picture of the lake, providing a suitable reference for further chemical and biological comparisons. Subsequently, the Institute of Botany, University of Padua, made limnological investigations on the lake in 1971–1972 and 1976–1977. Their phytoplankton production estimate (oxygen method) is useful for determining the trophic state of the lake. Their monthly data for the period from March to July of these years range (transformed into carbon) from 1.0–1.9 to 1.3–2.9 g C m^{-2} day^{-1}, for net and gross production, respectively. A_{max} for gross production is in the range of 0.19–0.43 g C m^{-3} day^{-1}. Further indications are also given by the low values of the ratio $\Sigma A/A_{max}$ (range: 3.5–6.9; see Fig. 4). These data, though limited to the spring–summer period, are enough to classify the lake as very productive (unpublished data by Paganelli et al.). The lake is shifting to eutrophy rather quickly, as shown by the decrease in transparency values from 1971–1972 to 1976–1977 (Fig. 4; Cordella et al. 1977; other unpublished data from the same authors).

Investigations on lake phytoplankton are being made, but little has been published so far.

In 1971–1972 diatoms and green algae prevailed, followed by the blue-greens. *Oscillatoria tenuis* was first reported in the lake in 1976 (Paganelli, personal communication). No regular zooplankton investigation has been made. We know, however, that, as in Lake Garda, *Eudiaptomus steueri*

TABLE 6

Mean Temperatures and Main Concentrations

	L. ISEO				
	1973	1974	1975	1976	1977
Temperature, °C	6.0	6.1	6.3	6.3	6.5
Oxygen, mg O_2 l^{-1}	6.46	5.31	5.29	3.42	3.90
Oxygen, % sat.	54.9	45.4	45.7	29.7	33.5
pH	7.80	7.80	7.75	7.85	7.91
Nitrate, μg N l^{-1}	700	665	575	730	755
Reactive phosphorus, μg P l^{-1}	28	30	47	58	46
Total phosphorus, μg P l^{-1}	35	35	47	65	49
Reactive silica, mg Si l^{-1}	1.96	1.41	1.30	2.56	2.07

	L. COMO				
	1973	1974	1975	1976	1977
Temperature, °C	6.2	6.2	6.3	6.4	6.5
Oxygen, mg O_2 l^{-1}	6.66	5.74	5.62	4.74	4.42
Oxygen, % sat.	57.0	49.1	48.5	40.8	38.2
pH	7.57	7.42	7.42	7.50	7.42
Nitrate, μg N l^{-1}	779	720	711	678	722
Reactive phosphorus, μg P l^{-1}	66	75	77	71	82
Total phosphorus, μg P l^{-1}	70	80	79	88	90
Reactive silica, mg Si l^{-1}	1.73	1.12	1.28	2.58	2.36
Ammonia, μg N l^{-1}					
Inorganic nitrogen, μg N l^{-1}					

replaces *E. padanus*, which is present in the other western deep lakes and that *Mesocyclops leuckarti* and *Mixodiaptomus laciniatus* seem to be absent (Baldi 1935, Pirocchi 1944, Bonomi and Gerletti 1967).

The profundal benthos has been sampled in 1967 (Bonomi and Gerletti 1967) and in 1972–1973 (Bonomi, unpublished data).

Biotic diversity is lower than in Lake Garda (Fig. 5), but population densities are much higher.

Lake Como

Lake Como is the deepest (410 m) lake in Western Europe.

Its southern section diverges into two basins which are different both morphologically and hydrobiologically (Fig. 6). Water renewal is to be considered more effective in the shallower Lecco basin, where lake water outflows through the River Adda. The very deep SW branch is practically a separate basin because of the presence of an underwater ridge between the Bellagio peninsula and the opposite western coast (maximum depth: 139 m). The hydrographic segregation, the human pressure exerted by the town of Como and very modest water inflows are the major causes of the higher trophy of this sub-basin.

Thermal and chemical data for 1973–1977, concerning Station 1 (Fig. 6), are given in Table 7. In Table 8, mean epi- and hypolimnetic values (March 1977) for stations 1 to 4 (Fig. 6) are tabulated.

of the Main Chemical Variables in Waters below 200 m

L. GARDA					L. MAGGIORE				
1973	1974	1975	1976	1977	1973	1974	1975	1976	1977
7.7	7.9	7.9	8.1	8.1	5.9	6.0	6.1	6.2	6.2
9.28	9.00	8.46	8.41	9.25	7.95	7.10	6.76	6.11	5.58
80.7	79.0	7.40	74.2	81.7	67.5	60.4	57.6	52.4	47.6
8.02	8.13	7.95	8.00	8.10	7.30	7.28	7.24	7.21	7.20
355	344	331	332	363	744	730	772	759	777
12	6	11	9	5	22	20	20	38	39
—	8	13	15	15	—	33	31	40	49
1.18	0.83	0.78	1.46	0.98	1.39	1.16	1.25	1.62	1.62

L. LUGANO			
1974	1975	1976	1977
5.3	5.4	5.4	5.4
0.15	0.43	0.03	0.00
1.5	3.7	0.2	0.0
7.65	7.65	7.60	7.60
40	64	—	25
147	152	159	202
162	166	178	204
1.17	1.87	2.49	2.06
253	330	—	443
293	394	—	468

Data in Table 7 indicate the absence of a complete overturn for the deepest station during the whole observation period. The oxygen concentration below 200 m has decreased (6.7 to 4.4 mg l^{-1}; Table 6), consistent with an accumulation of nutrients: 66 to 82 μg $P-PO_4$ l^{-1} and 70 to 90 μg total-P l^{-1}.

Our data together with those of Vollenweider (1965) and Tonolli et al. (1975) indicate an increase of phosphate concentrations, from 20 μg l^{-1} in 1960–1962, to 38 μg in 1971 (April) and up to 73 μg in 1977 (March). Nitrates, in the same period, have increased: from 500–550 (Vollenweider 1965) to 660 (Tonolli et al. 1975) and 755 μg N l^{-1} in 1977.

In Table 8 a comparison between the lake sub-basins is given. It is evident that Station 2 reflects the worse condition of the lake, while Station 3 is intermediate between Stations 2 and 4.

The above description of the peculiar features of the SW sub-basin, in particular the Como bay, also explains an earlier degradation in this section of the lake.

Monti (1925) mentioned a first bloom of *Microcystis aëruginosa* in August 1925, and Baldi (1949) described a *Fragilaria–Ceratium–Ulothryx* bloom that developed in September 1946. Phytoplankton countings have only recently been undertaken on a material collected in 1970–1971 (Braga 1972). The results demonstrate that blue-green algae are the most important components of the phytoplankton community, *Oscillatoria rubescens* and *Gomphospheria lacustris* being the dominant species. Diatoms, mainly represented by *Fragilaria crotonensis*, are not present in large numbers,

TABLE 7

Temperature and Main Chemical Variables of Lake Como

Depth	17/4 1973	28/3 1974	27/3 1975	30/3 1976	21/3 1977	17/4 1973	28/3 1974	27/3 1975	30/3 1976	21/3 1977
	Temperature (°C)					pH				
0	7.7	8.6	8.3	8.5	7.2	7.7	8.2	8.2	8.2	7.8
10	7.1	7.6	7.1	7.6	—	7.8	7.9	8.0	7.8	—
30	7.1	6.8	6.9	7.0	7.0	7.8	7.8	7.9	7.8	7.7
50	7.0	6.7	6.9	6.8	6.7	7.8	7.7	7.8	7.6	7.7
100	6.9	6.6	6.7	6.6	6.7	7.8	7.6	7.6	7.6	7.7
150	—	—	6.6	—	6.6	—	—	7.5	—	7.6
200	6.3	6.4	6.5	6.6	6.5	7.6	7.5	7.5	7.5	7.6
300	6.2	6.2	6.3	6.4	6.5	7.6	7.4	7.4	7.5	7.4
400	6.2	6.2	6.3	6.4	6.5	7.5	7.4	7.4	7.5	7.3
	Oxygen (mg l^{-1})					Oxygen (% sat.)				
0	10.5	13.0	12.9	12.7	10.6	98	118	117	115	93
10	10.5	11.0	11.2	10.7	—	92	97	98	95	—
30	10.3	9.2	10.5	10.0	9.0	90	80	92	87	79
50	10.3	8.7	9.9	8.9	8.5	90	75	86	77	74
100	9.7	8.1	8.0	8.5	8.1	84	70	69	73	70
150	—	—	7.8	—	7.6	—	—	68	—	66
200	7.3	7.2	7.3	6.6	6.2	63	62	63	57	54
300	6.5	5.7	5.1	4.5	4.5	56	49	44	39	39
400	6.4	4.6	5.1	3.6	2.8	55	40	44	31	24
	Nitrate (μg N l^{-1})					Silicate (mg Si l^{-1})				
0	670	560	610	540	690	1.3	0.6	0.6	1.3	1.5
10	670	690	660	630	—	1.3	0.6	0.7	1.6	—
30	670	810	700	680	820	1.4	0.8	0.8	1.7	1.6
50	800	810	680	710	810	1.3	1.0	0.9	1.8	1.7
100	730	720	730	710	830	1.3	0.9	1.1	1.8	1.7
150	—	—	710	—	780	—	—	0.9	—	1.8
200	780	720	770	610	720	1.7	0.9	1.0	2.0	1.8
300	820	720	700	710	730	1.7	1.0	1.4	2.8	2.2
400	710	720	680	680	710	1.8	1.5	1.3	2.7	3.1
	Reactive phosphorus (μg P l^{-1})					Total phosphorus (μg P l^{-1})				
0	39	23	32	18	39	44	26	49	34	58
10	42	47	40	28	—	49	49	52	51	—
30	42	64	45	34	61	48	74	49	55	71
50	42	59	48	45	67	50	64	54	55	74
100	53	52	58	51	66	62	62	58	60	76
150	—	—	58	—	69	—	—	58	—	72
200	69	63	59	54	58	80	72	64	83	61
300	64	81	80	75	87	66	82	81	88	95
400	68	74	87	80	93	70	82	87	91	105

while green algae, together with the blue-greens, represent the bulk of the Lake Como phytoplankton.

Primary production has been estimated at Stations 1, 3 and 4 for a total of 11 experiments. The highest value, 3010 mg C m^{-2} day^{-1}, was measured at Station 1, while the others ranged from 260 to 1820 mg C m^{-2} day^{-1}. Data reported here, together with the $\Sigma A/A_{max}$ values (Fig. 4) describe the lake as very productive.

Profundal benthos was sampled in all the lake basins along three sampling stretches (Fig. 6). The high population densities in the northern stations

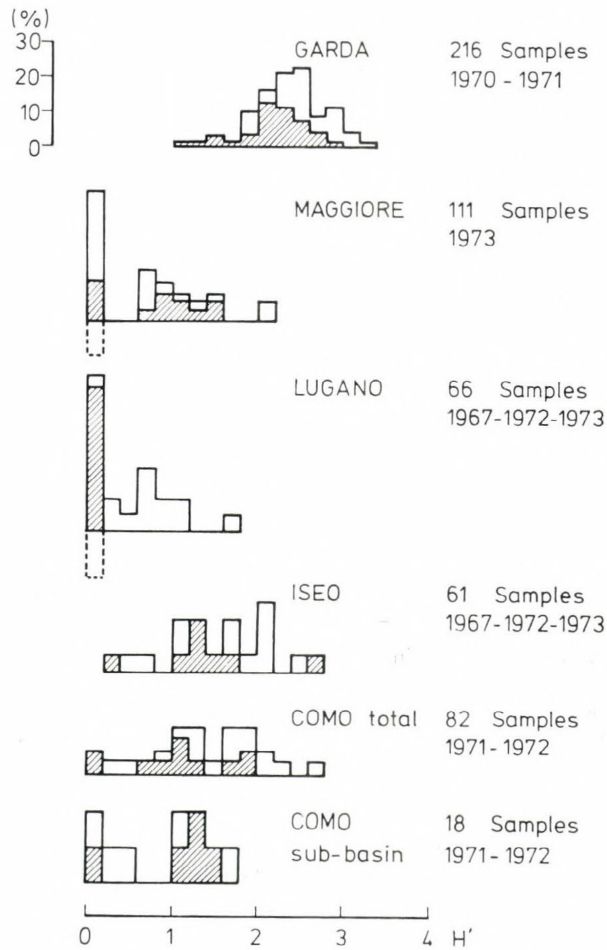

Fig. 5. Biotic diversity (H') frequency distribution of benthonic communities between −50 m and the maximum depth. Cross-hatched areas: −200 m to maximum depth; dashed areas (Lakes Maggiore and Lugano): absence of macrobenthos

are probably related to the large inflows from the main lake tributary (Fig. 5). Biotic diversity values are somewhat lower than in Lake Iseo. The Como sub-basin represents the most stressed situation in the whole lake (Fig. 5).

In contrast, a population of *Niphargus foreli* was found in the SE basin, at a depth of 200 m.

Lake Lugano

This is a morphologically complicated lake, articulated in many basins (Fig. 6), the largest and deepest of which is the north-eastern one ($z_{max} = 288$ m). More detailed morphometry may be found in Baldi et al. (1949).

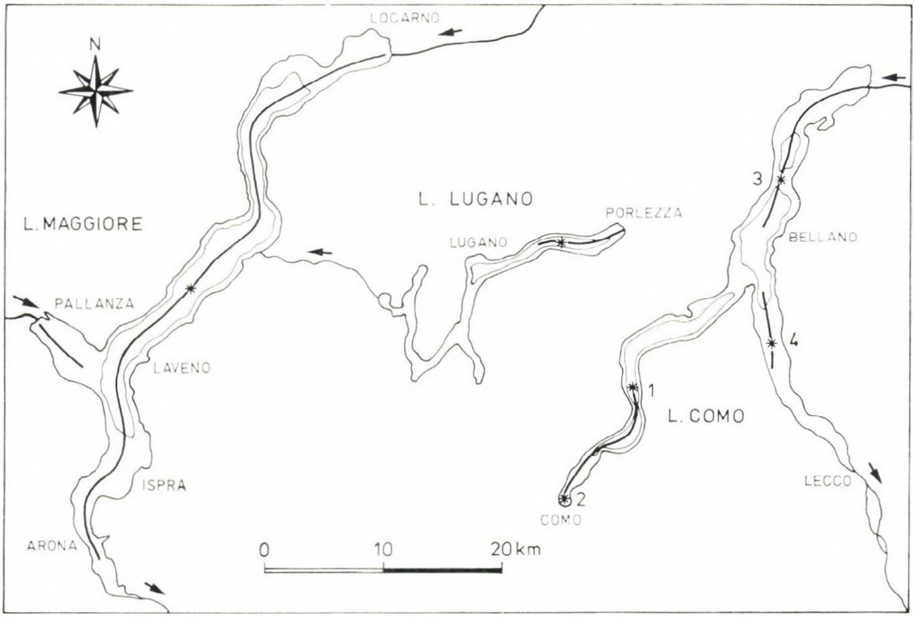

Fig. 6. Water sampling stations and benthos sampling profiles

We established our chemical and biological sampling stations in the NE basin (Fig. 6) which is morphologically very close to the other deep marginal lakes.

TABLE 8

Mean Values in Epilimnion (0–30 m, First Line) and Hypolimnion (30 m — Bottom, Second Line) at Four Stations in Lake Como (see Fig. 6)

	Station 1 (—410 m)	Station 2 (—83 m)	Station 3 (—280 m)	Station 4 (—183 m)
pH	7.87	7.61	7.80	7.95
	7.54	7.59	7.70	7.74
Oxygen, mg l^{-1}	10.91	9.29	10.60	11.40
	6.24	7.99	9.02	9.24
Oxygen, % sat.	95.8	79.6	90.1	97.4
	53.7	67.5	75.4	77.3
Ammonia, μg N l^{-1}	49	63	27	33
	6	39	12	9
Nitrate, μg N l^{-1}	630	760	660	610
	720	780	680	690
Reactive phosphorus, μg P l^{-1}	28.7	78.7	40.4	27.8
	61.7	78.6	42.3	41.2
Total phosphorus, μg P l^{-1}	50.6	99.9	50.1	48.8
	77.7	93.6	49.1	52.0
Reactive silica, mg Si l^{-1}	1.60	1.84	1.74	1.62
	2.38	1.90	1.93	1.82

Our results confirm a striking oxygen deficit of this lake: the waters below 150 m, with extremely low oxygen concentrations in 1974 and 1975, were practically anoxic in the following two surveys (Table 9). Nitrates peak (600–670 μg N l^{-1}) in the 20–50 m layer; in deep water the $N-NO_3$ concentration drops and ammonia attains values as high as 400–500 μg N l^{-1}. Reactive and total phosphorus in deep layers exceed concentration values of 200 μg P l^{-1}. Both ammonia and phosphate deep-water concentrations are the highest found in all lakes we have considered.

An attempt to describe the evolution of this lake, in comparison with previous available data, gives a really alarming picture of the situation. Baldi et al. (1949) could detect a starting point of eutrophication in 1946–1947: they found that the water below 125 m had already developed a strong oxygen deficit, the concentration being at that time in the range of 2.4 to 3.8 mg O_2 l^{-1}. During further investigations (Vollenweider 1965, Jaag and Märki 1970), the oxygen concentrations in the deepest layers remained extremely low, in the order of a few tenths of a mg per l. The oxygen reserve below 125 m, which was exhausted approximately in 1950, has not been recharged: it is evident, therefore, that a complete overturn of the lake does not exist.

The lake biota have drastically changed. Copepods are now represented in the zooplanktonic community only by *Cyclops abyssorum* and *Mesocyclops leuckarti*; Ravera (1973) reports, but not for the NE deep basin, the presence of *Acanthocyclops viridis*.

Already in 1949 Baldi et al. reported that *Eudiaptomus vulgaris* (= *padanus*), previously mentioned by Monti (1929), could not be found; they

TABLE 9

Temperature and Main Chemical Variables of Lake Lugano

Depth	29/3 1974	20/3 1975	2/4 1976	22/3 1977	29/3 1974	20/3 1975	2/4 1976	22/3 1977
	Temperature (°C)				pH			
0	9.4	6.1	9.8	7.4	8.6	8.0	8.7	8.7
10	7.0	5.8	7.0	—	8.0	8.0	8.3	—
20	—	5.8	6.2	5.7	—	7.9	8.1	8.0
30	5.7	5.8	5.8	—	7.9	7.8	7.9	—
50	5.5	5.6	5.6	5.6	7.8	7.8	7.8	7.8
100	5.4	5.5	5.6	5.5	7.6	7.6	7.6	7.6
150	5.3	5.4	5.4	5.5	7.6	7.7	7.6	7.6
200	5.3	5.4	5.4	5.4	7.7	7.7	7.6	7.6
250	—	5.4	5.4	—	—	7.7	7.6	—
280	5.3	5.4	5.4	5.4	7.6	7.5	7.6	7.6
	Oxygen (mg l^{-1})				Oxygen (% sat.)			
0	14.0	11.4	15.2	14.0	131	98	142	125
10	9.8	9.3	12.5	—	86	80	110	—
20	—	7.3	9.7	7.9	—	62	84	67
30	7.5	8.0	8.0	—	64	68	68	—
50	7.0	6.6	6.6	6.8	60	56	56	58
100	1.2	1.6	1.5	0.3	12	14	13	3
150	0.3	0.9	0.2	0.2	3	8	2	2
200	0.2	0.7	0.1	0.0	2	6	1	0
250	—	0.4	0.0	—	—	3	0	—
280	0.1	0.2	0.0	0.0	1	2	0	0

TABLE 9 *(cont'd)*

Depth	29/3 1974	20/3 1975	2/4 1976	22/3 1977	29/3 1974	20/3 1975	2/4 1976	22/3 1977
	Nitrate (μg N l^{-1})				Silicate (mg Si l^{-1})			
0	410	590	250	230	0.2	0.5	0.4	0.3
10	530	600	400	—	0.2	0.6	0.4	—
20	—	640	440	600	—	0.6	0.7	0.9
30	670	660	510	—	0.5	0.7	0.8	—
50	630	660	590	620	0.5	0.6	1.0	0.9
100	400	430	390	310	0.9	1.2	1.4	1.4
150	120	90	30	30	0.9	1.5	2.0	1.7
200	40	80	10	20	0.9	1.7	2.2	1.9
250	—	50	10	—	—	1.9	2.6	—
280	40	70	10	30	1.4	2.0	2.6	2.2
	Reactive phosphorus (μg P l^{-1})				Total phosphorus (μg P l^{-1})			
0	61	78	58	46	76	101	95	98
10	84	88	76	—	89	106	103	—
20	—	89	91	96	—	104	108	106
30	94	118	99	—	105	159	116	—
50	95	97	105	104	102	108	115	110
100	98	97	113	121	105	103	117	129
150	100	96	114	135	124	104	123	143
200	124	135	99	177	154	143	156	182
250	—	155	177	—	—	176	183	—
280	166	164	193	222	168	174	193	222
	Ammonia (μg N l^{-1})				Inorg. nitrogen (μg N l^{-1})			
0	61	15	—	116	480	610	—	350
10	44	19	—	—	580	620	—	—
20	—	7	—	15	—	650	—	620
30	6	7	—	—	680	670	—	—
50	0	7	—	12	630	670	—	630
100	0	4	—	17	400	430	—	330
150	20	67	—	200	140	160	—	230
200	15	226	—	327	60	310	—	350
250	—	352	—	—	—	400	—	—
280	451	406	—	540	490	480	—	570

sampled rich populations of *Mixodiaptomus laciniatus*, now completely absent. Consequently, the diaptomids seem to have completely disappeared from the zooplankton. Ravera (1973) quotes a personal communication by O. Jaag, from which it is apparent that these entomostracans have practically been absent in the lake since at least 1956. Among the cladocerans, *Daphnia obtusa* seems to be a new settler of the last decade; it apparently thrives only in the minor SW shallower basins. Since 1956, on the contrary, *Sida crystallina* and *S. limnetica* have disappeared (Ravera 1973).

In the period of 1944–1945, phytoplankton was dominated by the diatoms *Asterionella formosa* and *Fragilaria crotonensis*, followed, in order of importance by the greens, the blue-greens and the flagellates. In 1955, the presence of *Oscillatoria rubescens* at 'all sampling points' was reported (Jaag and Märki 1970); this alga was recorded in the lake since 1945. In the subsequent years the winter plankton was dominated by *O. rubescens* (Jaag and Märki 1970). Ravera (1973) reports, however, the explosion of a massive summer (July) bloom of this blue-green all over the lake.

In spite of the quantity of data collected in these last years by Ravera and his collaborators, the data published on phytoplankton production are scanty (Premazzi et al. 1976): we know that a production of up to 5 g C m^{-2} day^{-1} has been measured in the NE basin; this, together with the very low values of the $\Sigma A / A_{max}$ ratio which can be calculated from their data, indicate the high productivity of this lake.

The profundal benthos (sampling profile in Fig. 6) is consistent with environmental data: diversity and population density are rather low (Fig. 5) and rapidly decrease with depth.

Lake Maggiore

Lake Maggiore (Fig. 6) was formed by two glaciers, the first descending through the Ticino Valley, which is the deepest ($z_{max} = 370$ m) and longest valley; the second one, from the Ossola Valley, which joined the first orthogonally, and is responsible of the formation of the shallower ($z_{max} = 150$ m) Pallanza Bay. Lake Maggiore is the one investigated most among the largest subalpine lakes; for more detailed bibliographical references, the reader is referred to Bonomi (1968), Bonomi et al. (1970) and Barbanti et al. (1974).

This paper presents more recent information from the last five years and other, partly unpublished, data concerning the period between 1964 and 1977. The sampling point for chemical analyses and the line for profundal benthos sampling are shown in Fig. 6.

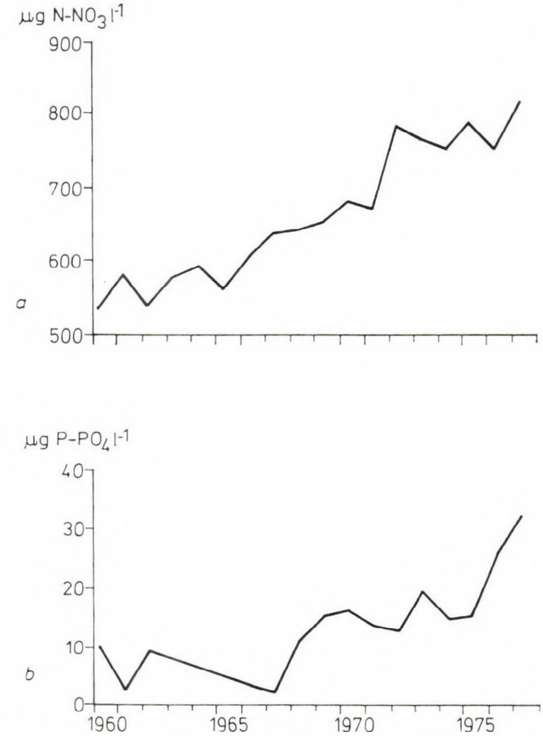

Fig. 7. Nitrate (*a*) and reactive phosphorus (*b*) trends in Lake Maggiore: mean concentration in the whole water column

TABLE 10

Temperature and Main Chemical Variables of Lake Maggiore

Depth	12/3 1973	4/4 1974	21/3 1975	24/3 1976	1/4 1977	12/3 1973	4/4 1974	21/3 1975	24/3 1976	1/4 1977
	Temperature (°C)					pH				
0	6.6	10.3	6.6	8.1	6.9	7.4	8.3	7.4	7.9	7.5
5	6.3	8.2	6.6	7.2	—	7.5	7.8	7.5	7.7	—
10	6.3	7.3	6.6	7.0	6.7	7.5	7.6	7.5	7.7	7.5
20	6.3	6.9	6.6	6.6	—	7.5	7.5	7.5	7.5	—
30	6.2	6.8	6.5	6.3	6.7	7.5	7.5	7.5	7.5	7.5
50	6.2	6.4	6.5	6.3	6.5	7.5	7.3	7.4	7.5	7.5
100	6.2	6.3	6.3	6.3	6.4	7.4	7.3	7.3	7.4	7.4
150	6.1	6.1	6.2	6.3	6.3	7.3	7.4	7.3	7.3	7.3
200	6.0	6.1	6.2	6.2	6.2	7.3	7.3	7.3	7.3	7.2
250	5.9	6.0	6.1	6.2	6.2	7.3	7.3	7.3	7.2	7.2
300	5.9	6.0	6.1	6.2	6.2	7.3	7.3	7.2	7.2	7.2
360	5.9	6.0	6.1	6.2	6.2	7.3	7.2	7.2	7.2	7.2
	Oxygen (mg l^{-1})					Oxygen (% sat.)				
0	10.7	13.8	10.5	12.6	10.5	92	131	91	113	91
5	10.7	12.6	10.5	11.8	—	92	113	91	104	—
10	10.7	11.1	9.9	11.4	10.1	92	98	86	95	88
20	10.7	10.4	10.4	10.1	—	92	86	90	87	—
30	10.7	9.9	9.9	9.9	9.6	92	86	86	85	83
50	10.7	9.8	10.2	9.7	9.4	92	84	86	84	81
100	9.4	8.9	7.7	9.4	8.7	80	77	66	81	75
150	8.3	8.6	7.3	7.8	6.5	71	74	63	67	55
200	8.1	8.0	7.3	7.4	6.2	69	68	63	64	53
250	8.1	7.3	7.3	6.6	5.6	69	62	62	57	48
300	8.0	7.1	6.2	5.6	5.4	68	61	53	48	46
360	7.6	6.3	6.5	5.4	5.4	64	54	56	47	46
	Nitrate (μg N l^{-1})					Silicate (mg Si l^{-1})				
0	740	710	720	670	780	0.8	0.7	0.7	0.9	1.4
5	750	660	720	420	—	0.9	0.6	0.7	1.0	—
10	760	730	720	740	790	0.9	0.6	0.6	1.0	1.4
20	770	780	770	770	—	0.9	0.6	0.6	1.2	—
30	770	780	760	770	860	0.9	0.6	0.6	1.2	1.4
50	770	800	830	770	850	0.9	0.6	0.8	1.3	1.4
100	770	760	840	780	870	1.0	0.8	0.7	1.3	1.5
150	810	780	770	690	840	1.3	0.8	1.0	1.5	1.6
200	770	730	800	760	800	1.3	1.0	1.0	1.6	1.6
250	740	740	740	770	790	1.4	1.0	1.2	1.6	1.6
300	740	700	740	750	790	1.4	1.2	1.3	1.6	1.6
360	740	760	840	760	730	1.4	1.4	1.4	1.7	1.7
	Reactive phosphorus (μg P l^{-1})					Total phosphorus (μg P l^{-1})				
0	16	7	3	9	16		21	20	23	23
5	21	9	2	7	—		23	18	19	—
10	13	12	2	7	18		25	16	14	30
20	15	3	3	7	—		20	19	15	—
30	19	2	5	11	24		19	14	16	32
50	10	7	5	11	20		18	17	18	28
100	21	8	13	16	21		17	24	22	33
150	16	15	15	9	34		23	28	29	40
200	19	17	16	32	32		28	21	35	44
250	15	17	13	37	39		31	32	39	50
300	26	19	21	37	39		35	35	39	50
360	28	25	29	44	42		35	32	45	50

As the lake is oligomictic, it is important to refer the chemical and thermal data to the sequence of holomictic-meromictic years. We know that in Lake Maggiore the two last complete overturns occurred in 1963 (Vollenweider 1964, Tonolli and Bonomi 1967) and 1970 (Barbanti et al. 1974).

The deep water temperature gradually increased from 5.9 (1973) to 6.2 °C in 1977 (Table 10). Oxygen concentration below 200 m (Table 6) shows a regular decrease from 7.95 to 5.58 mg l^{-1}, consistent with an accumulation of reactive phosphorus (22 to 39 μg P l^{-1}), total phosphorus (33 to 49 μg P l^{-1}) and nitrates (740 to 780 μg N l^{-1}), in the five year period. The trend of nitrate concentrations is shown in Fig. *7a* for the entire period 1960–1977; the mean rate of increase (530 to 812 μg N l^{-1}) is estimated as 16.5 μg N l^{-1} yr^{-1}.

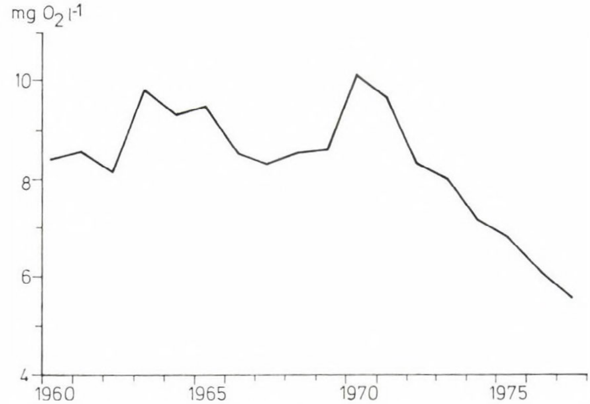

Fig. 8. Oxygen trend in Lake Maggiore: mean concentration in the water column below 200 m

Reactive phosphorus has been regularly analysed since 1966; previous measurements were made at rather irregular intervals. The average accumulation rate for 1966–1977 is 2.6 μg P l^{-1} yr^{-1}, but the progression is striking, because, while the 1968–1975 values are still in the range of 10–20 μg l^{-1}, the concentration for 1976 and 1977 increases up to 25 and 32 μg P l^{-1}, respectively (Fig. *7b*).

The pluriannual trend of oxygen concentration below 200 m (Fig. 8) reveals two peaks: in 1963 and 1970, resulting from the two full overturns. The corresponding mean oxygen concentrations were 9.7 and 10.0 mg l^{-1}, while the whole water column attained 82–84 per cent saturation. The non-saturation of the large insubrian lakes had been noticed by Vollenweider (1965), whose interpretation was that the undersaturation might be due to the short mixing period in relation to the rate of 're-charge' of oxygen in the lake waters.

From Fig. 8 it is also evident that in the two periods following the 1963 and 1970 circulations, the average rates of oxygen depletion were quite different: 0.20 mg O_2 l^{-1} yr^{-1} in 1963–1969, and three times higher (0.64 mg O_2 l^{-1} yr^{-1}) in 1970–1977.

The values of primary production calculated over a long period of time (starting from 1954) are in good agreement with the data of oxygen depletion shown in the following table (Ruggiu and Saraceni 1977):

	$g\ C\ m^{-2}\ yr^{-1}$
1954–1961	115–175
1965	290
1970–1971	370
1973–1974	387

Despite the fact that primary production seems to have been practically constant in recent years (yearly values), the sequence of the $\Sigma A/A_{max}$ ratios indicates a progressive eutrophication (Fig. 4). The values shift in fact from an average of 11.2 in 1959–1960 (Vollenweider 1960) to 9.8 in 1970–1971 and 7.6 in 1973–1974 (Ruggiu and Saraceni 1977).

The phytoplankton community has changed substantially during the last fifteen years. The first bloom of *Tabellaria flocculosa* in 1962 was followed by the appearance — for the first time over the whole lake — of *Oscillatoria rubescens* in 1967 (Bonomi et al. 1970). The phytoplankton is now characterized by a general dominance, in all seasons by blue-green algae (mainly *O. rubescens*, *Microcystis aëruginosa* and *Lyngbya limnetica*). The modifications in the zooplankton community structure are also remarkable. A decrease in the per cent density of the cyclopids *Mesocyclops leuckarti* and *Mixodiaptomus laciniatus* is coupled with an increase of *Daphnia hyalina*. *Sida crystallina* and *Heterocope saliens* have disappeared, while *Chydorus sphaericus* has been found with high pelagic densities for some years (Bonacina 1977).

Additional information on the modifications of Lake Maggiore communities derive from the profundal benthos, with the disappearance of the amphipod *Niphargus foreli*, together with the decline of *Peloscolex ferox* and *Bythonomus lemani* and the first records of *Tubifex tubifex*, *Potamothrix hammoniensis* and *Stylodrilus heringianus* (Bonomi 1968 and 1969). A recent (1973) survey along the axis of the lake (Fig. 6) supports the evidence of the modification of this community. In fact the average diversity values are rather low and large areas of the profundal zone are completely devoid of macrobenthos.

DISCUSSION

In oligomictic lakes the chemical characteristics of the deep water layers not involved in the full circulation are the consequence not only of the trophic condition of the year, but also of the time lag from the last complete overturn. These layers, in other words, accumulate the result of pluriannual sinkings; the resulting breakdown of the organic matter leads to accumulation of nutrients, mainly phosphate, and to a corresponding oxygen depletion. Concentration changes in such segregated waters can be suitably used both for comparison between lakes and the detection of their evolution.

On the basis of our own and previous observations (Vollenweider 1964) we estimate that the 200 m depth is the lowest limit which may be reached by the vertical mixing, when a full circulation does not occur. The mean temperatures and the chemical features in the waters below 200 m are reported in Table 6.

The data show the variation patterns of reactive phosphorus, total phosphorus and dissolved oxygen. It is evident that Lake Garda and Lake Lugano represent two extreme situations: the first one exhibits favourable oxygenation levels and moderate and relatively constant phosphate concentrations; the second shows oxygen concentration of only tenths of a mg in 1974 and 1975 and practically zero in 1976–1977. In Lake Lugano the already high reactive and total phosphorus concentrations are still increasing.

The deepest zone of the other lakes shows similar qualitative trends: the rates of oxygen depletion and of reactive and total phosphorus increase are reported in Table 11. The figures have been obtained by subtracting the

TABLE 11

Oxygen Decrease, Reactive and Total Phosphorus Increase in the Water Column below 200 m. The Mean Yearly Value Has Been Calculated for the Years Reported in the Table

	L. MAGGIORE 1973–1977	L. COMO 1973–1977	L. ISEO 1973–1976	L. LUGANO 1974–1977	L. GARDA 1973–1976
Oxygen decrease, $\mu g\ O_2\ l^{-1}\ yr^{-1}$	590	570	1010	—	270
Reactive phosphorus increase, $\mu g\ P\ l^{-1}\ yr^{-1}$	4.1	3.9	9.9	18.3	—
Total phosphorus increase, $\mu g\ P\ l^{-1}\ yr^{-1}$	5.4	5.6	9.9	14.0	—
Annual areal oxygen depletion below 200 m, $g\ O_2\ m^{-2}\ yr^{-1}$	63.6	71.9	40.0	—	22.8

1973 value from that of 1977; for Lakes Garda and Iseo the 1973–1976 difference has been used, because of the already mentioned partial deep water renewal in 1977. Lake Iseo shows the highest oxygen depletion rate; the phosphate increase in this lake is only exceeded by that of Lake Lugano. These data seem to indicate that Lake Iseo is very productive, but cannot be used for a direct comparison with the other lakes; indeed, the water below 200 m in this lake represents a smaller volume than that of the other lakes examined (see Table 1).

The high reactive and total phosphorus accumulation rates in Lake Lugano can be related to anoxic release from sediments and to the different nutrient regeneration mechanisms. While for Lakes Maggiore, Como and Iseo it may be assumed that reactive and total phosphorus mainly result from aerobic decomposition processes, in Lake Lugano the regeneration probably attains a higher rate because of the anoxic conditions.

As we have seen, the oxygen decrease and the phosphate accumulation rates are similar for Lake Como and Lake Maggiore.

If we take into account the greater depth of Lake Como, we conclude that this has to be considered more productive than Lake Maggiore. In fact, if we multiply the oxygen decrease by the volume below 200 m and divide it by the area of the 200 m isobath, and then express the rate, in analogy with the classic hypolimnetic deficit, as mg O_2 m^{-2} yr^{-1}, we obtain the result shown in Table 11. The area and volume used for calculations on Lake Como are only those of the Como sub-basin. Assuming that this index, i.e. the flow of oxygen through the -200 m area may counterbalance the mean yearly oxygen depletion for the different lakes, the following succession seems to exist: Lakes Como, Maggiore, Iseo, Garda. This can be used as a rough index of the level of eutrophication which has been reached in the single lakes. The annual areal oxygen depletion below 200 m (Table 11) suggests for Lake Como (Como sub-basin) a condition worse than for Lake Maggiore. It must be remembered, however, that the N and SE basins of Lake Como are in a better situation: it is, therefore, likely that Lake Maggiore and Lake Como are trophically closer if their average conditions are considered.

The elaboration of benthos data are in agreement with the above described provisional conclusions. Lake Garda and Lugano appear again to be at extremes, and Iseo shows a better diversity frequency distribution in comparison with Lakes Como and Maggiore (Fig. 5). The data seem to indicate, however, that the condition in Lake Maggiore, which has a large area devoid of macrobenthos, is worse than in Lake Como.

These are the actual conditions of our large subalpine lakes. It has been stressed that conditions of the drainage basin, including man and his activities, play an extremely important role in accelerating the eutrophication processes, due to loading the lakes with different amounts of dissolved and particulate substances. These amounts usually depend on the number of inhabitants settled on each drainage basin and on the use of land.

TABLE 12

Actual and Permissible Phosphorus Loadings

	L. GARDA	L. ISEO	L. COMO	L. MAGGIORE	L. LUGANO (NE)
Inhabitants in the drainage basin	158 500	167 700	461 000	669 900	75 600
Phosphorus loading (probable range), g P m^{-2} yr^{-1}	0.31–0.62	1.98–3.96	2.31–4.62	3.4*	1.98–3.96
Critical phosphorus loading, g P m^{-2} yr^{-1}	0.31	0.91	1.06	1.31	0.61

* Directly measured.

It is well known that several nutrients may contribute to the increase of production in lakes. Nevertheless we do not want to enlarge on this problem here. For Lakes Maggiore, Como, Iseo and Garda, the ratio between nitrogen and phosphorus clearly indicates that the level of production in these lakes is controlled by phosphorus. This holds true also for Lake Lugano where the low N/P ratio is a direct consequence of the high eutrophication level. Unfortunately, direct measurements of phosphorus loading have only been obtained for Lake Maggiore, giving the result of 3.4 g m^{-2} yr^{-1}

(Calderoni and Mosello 1976). For the other lakes it has been possible to estimate the probable phosphorus loading range, assuming a daily phosphorus input varying from 2.0 to 4.0 g P $inhab^{-1}$ day^{-1} (all sources included). These results are summarized in Table 12, where the critical phosphorus loading according to Vollenweider's (1976) criteria is also given.

It can be seen that the actual phosphorus loadings for Lakes Iseo, Como, Maggiore and Lugano are much higher than the critical levels.

These data demonstrate quite clearly that measures taken to recover large lakes like these require much effort and great expenses because phosphorus input must be drastically reduced.

Acknowledgments. We are grateful to Prof. and Mrs. Arturo Paganelli, to Drs Paolo Cordella and Renata Trevisan at the Institute of Botany, University of Padua, for access to their unpublished data on Lake Iseo.

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SUMMARY

The large and deep lakes of the Italian subalpine district (Garda, Iseo, Como, Lugano and Maggiore, from East to West) are located in a region which has undergone considerable urban and industrial development over the last decades.

Their actual limnological conditions and their trophic state have been described in terms of their main physical, chemical and biological features. Also, a comparison between the conditions of the last five years and historical data has been made.

The results obtained from our research during spring time over the period 1973–1977 have demonstrated that these lakes are oligomictic. A full circulation occurred only in Lake Garda in April 1977, and Lake Iseo showed a partial renewal of the hypolimnic waters during the same period.

Lake Garda shows low nutrient content, high oxygen concentration in deep water, a moderate primary productivity and a high diversity index of the deep benthic communities. These characteristics are not substantially different from those described in the years before 1973.

Lakes Maggiore, Como (Como sub-basin) and Iseo, on the other hand, have reached a state of meso-eutrophy, and Lake Lugano (NE basin), strongly eutrophic for a long time, displays now worse conditions than in the past.

The production level of each lake has been evaluated on the basis of the rate of oxygen depletion and increase of reactive and total phosphorus concentrations in the deep, not regularly mixed hypolimnetic water masses.

Considering the degree of eutrophication, the lakes rank as follows: Garda $\ll$ Iseo $<$ Maggiore $<$ Como (Como sub-basin) $\lll$ Lugano (NE basin), which is consistent with values of primary productivity and biotic diversity.

The algal production of these lakes is controlled by phosphorus. The critical phosphorus loadings have been calculated according to Vollenwei-

der's (1976) criteria for the five lakes. Phosphorus inputs, directly measured for Lake Maggiore and estimated for the other lakes from the number of inhabitants in the drainage basin, are much higher than the critical loadings, except for Lake Garda.

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WATER QUALITY DETERIORATION AND RESTORATION IN POND 'SÓSTÓ' (NE-HUNGARY)

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Abstract

Water quality of Pond 'Sóstó' has undergone changes which are considered to have been caused by rapid cultural eutrophication. This process has recently advanced so much that proper water-quality conditions can only be restored by conscious human intervention.

INTRODUCTION

Pond 'Sóstó' near Nyíregyháza used to serve as a popular resort up to the 1960's. From this time on water quality deteriorated so rapidly that not only bathing has become impossible, but also the pond's aesthetic value decreased.

A detailed water analysis of Pond 'Sóstó' was carried out to characterize the alterations in water quality. A procedure had to be devised securing an effective but economical rehabilitation of the pond.

CHANGES IN LIMNOLOGICAL CHARACTER AND THE PRESENT STATUS OF POND 'SÓSTÓ'

Pond 'Sóstó' is situated in the north-eastern part of Hungary, near Nyíregyháza (Fig. 1). It is situated in a natural depression covered with meadow soil layers susceptible to sodiphication, deposited on upper Pleistocenic layers. The ground-water lies near the surface.

Originally, the pond was situated in a loamy-humic, impermeable layer capable of swelling. The present bed profile has formed during repeated deepenings. Since the earlier regular decrease of water level and the complete drying up of the bottom has stopped, the water cycle of the pond of an astatic nature has gradually assumed a semi-, and later on, a eustatic character. The pond has no inlets and surface outlets. The water is supplied from melt, precipitation and ground water. There is no intensive groundwater flow of a definite direction in the surroundings of the pond. However, a 'pulsating' type of water exchange seems to exist between the pond and the ground-water.

The surface of the catchment area of the pond is relatively small (8.2 ha) and well delimited (Fig. 2). Surface and ground-water pollution gets into the pond from the surroundings characterized by intensive horticulture and park-cultivation and from the strongly polluted Canal 'Igrice' situated on the eastern shore of the pond.

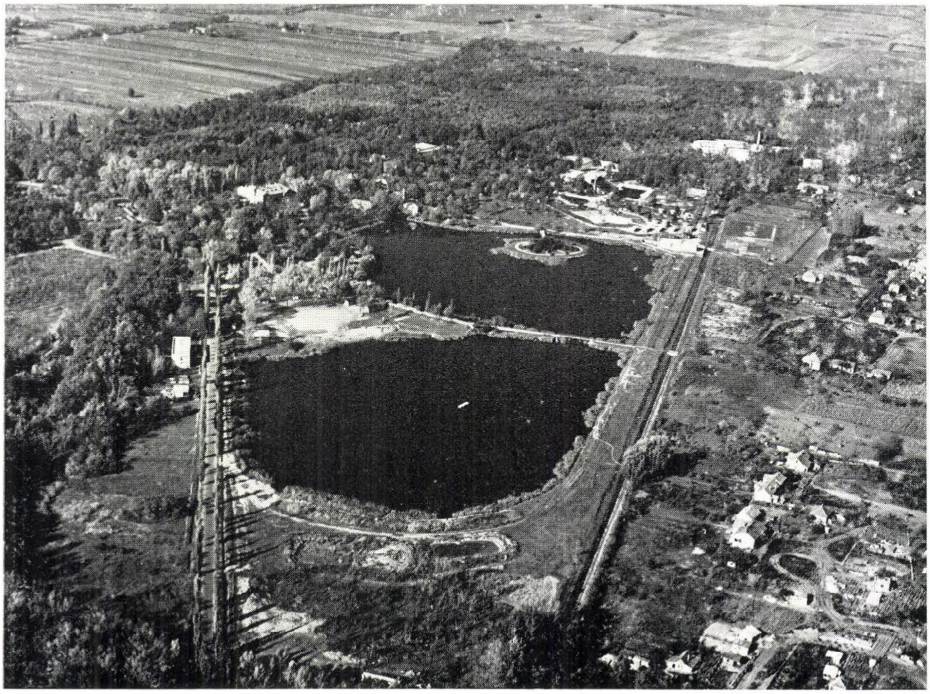

Fig. 1. Aerial view of Pond 'Sóstó' (Photo Járai, MTI)

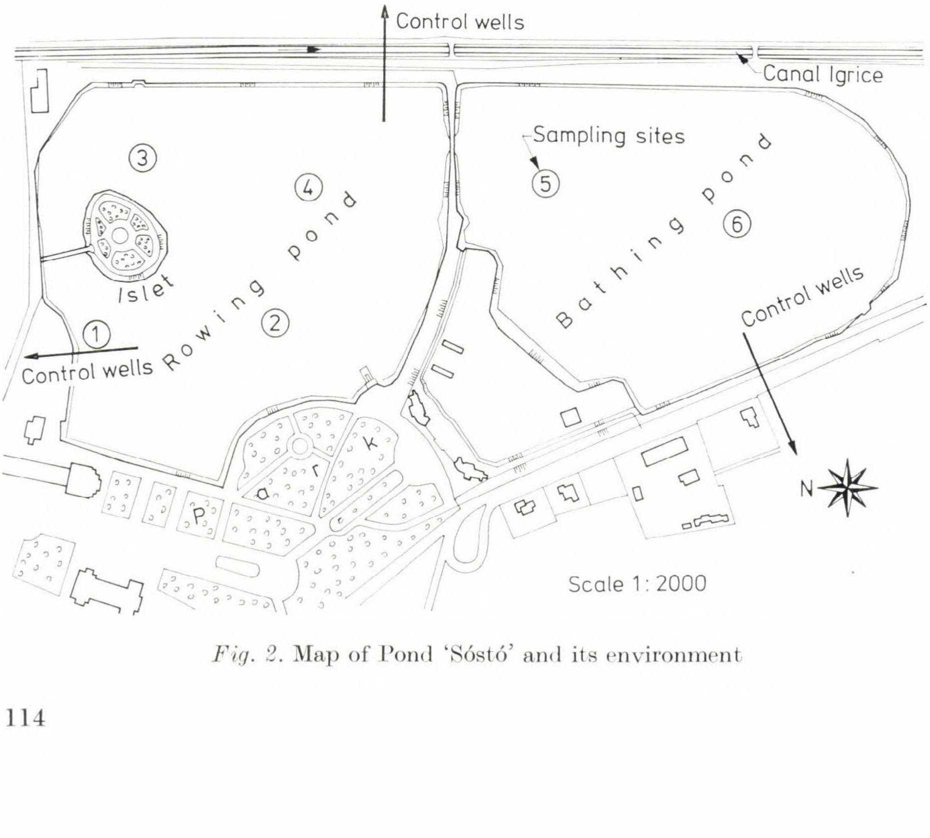

Fig. 2. Map of Pond 'Sóstó' and its environment

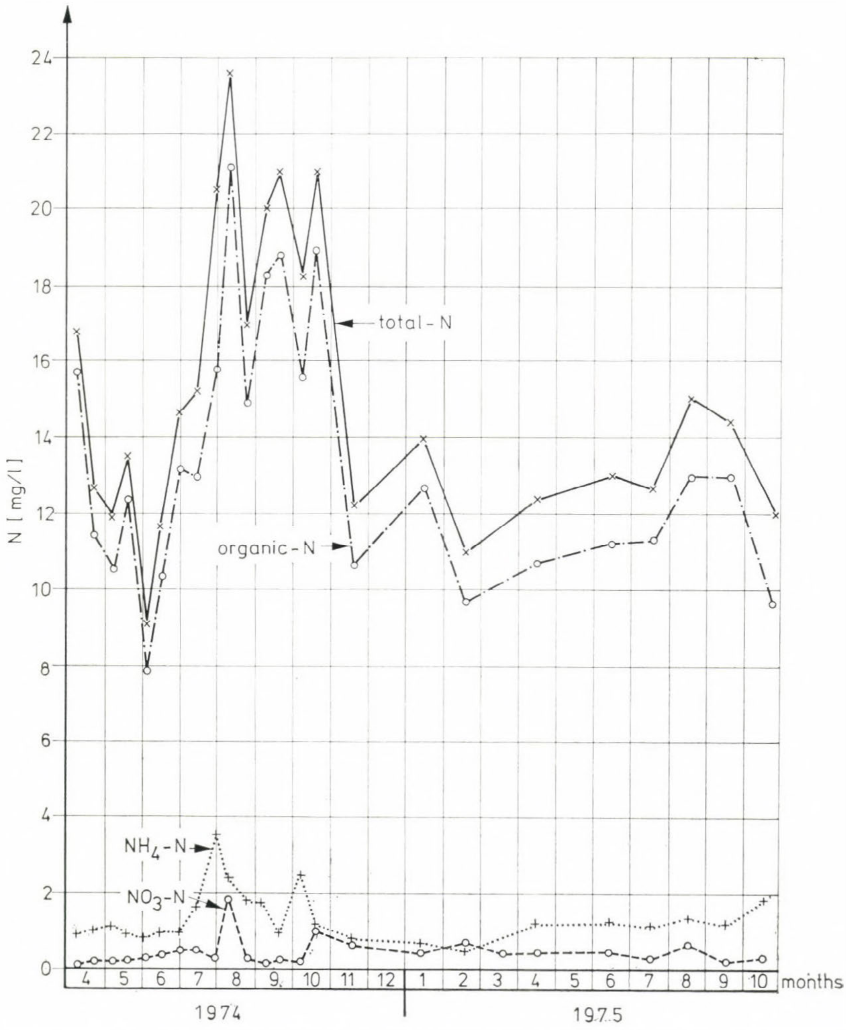

Fig. 3. Changes of various forms of N during 1974–1975

Pond 'Sóstó' is a shallow, stagnant water of about 1 m depth. It consists of two basins, which are almost same size. The southern one is used for bathing, the northern for rowing (Dévai 1976). The water level is 0.5–1 m below the surface of the surrounding area. Its shores are abrupt and crumbling, owing to the lack of a planned shoreline; consequently soil erosion is considerable.

The pond was dredged several times in order to prevent rapid siltation. But the 10–30 cm thick, loose mud layer could not be removed from the bottom, only the increased release of nutrients (especially P and N) was promoted. Chemically, the pond has been regarded as a soda or saline type water, but as a consequence of conducting the water of the heated, indoor swimming pool and the thermal water surplus of the neighbouring deep

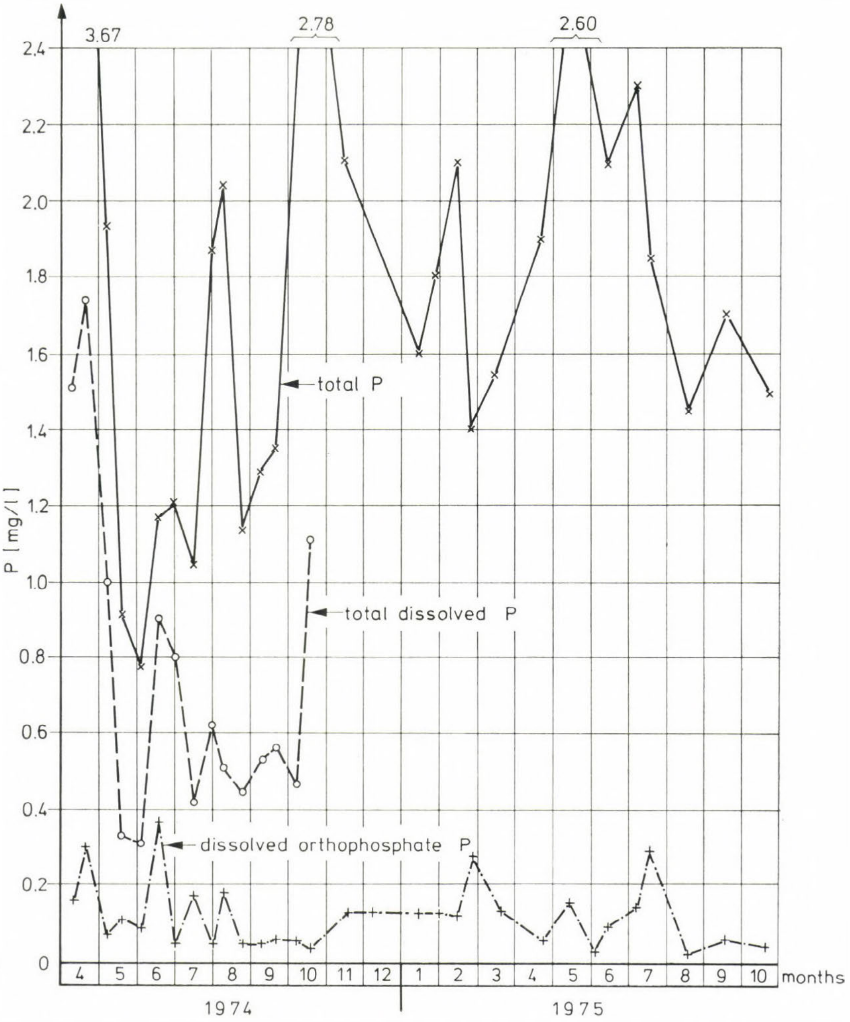

Fig. 4. Changes of various forms of P in the pond water during 1974–1975

wells into the pond between 1959–1965, the chemical composition of the water has been changing, first gradually and since 1959 rapidly. During this period the total salt (mainly NaCl) content increased threefold.

An about 3 to 10 m wide reed-belt borders the shore of the pond that was regularly burnt off during the last years.

Grass- and silver carps were introduced previously into the pond to increase fish populations for sport fishery and to reduce the biomass of algae and reed. These species did not survive because of the high salt content, high pH and significant concentration of ammonia, as well as the periodically occurring total anoxia.

In the parks and gardens encircling the pond huge amounts of various fertilizers and pesticides are used. They are washed into the water especially during heavy showers.

Water quality from the biological point of view is extremely bad (Figs 3, 4 and 5). According to Felföldy's (1974) classification, its figured code was 9970 in 1974–1975, i.e. polyhalobic, hypertrophic, alpha-meso-poly-saprobic and nontoxic. Deterioration of water quality was caused by rapid cultural eutrophication. This process cannot be stopped without organized human intervention (Edmondson 1970, Thomas 1971, Ohle 1973, Felföldy 1969, 1974, Dévai et al. 1976).

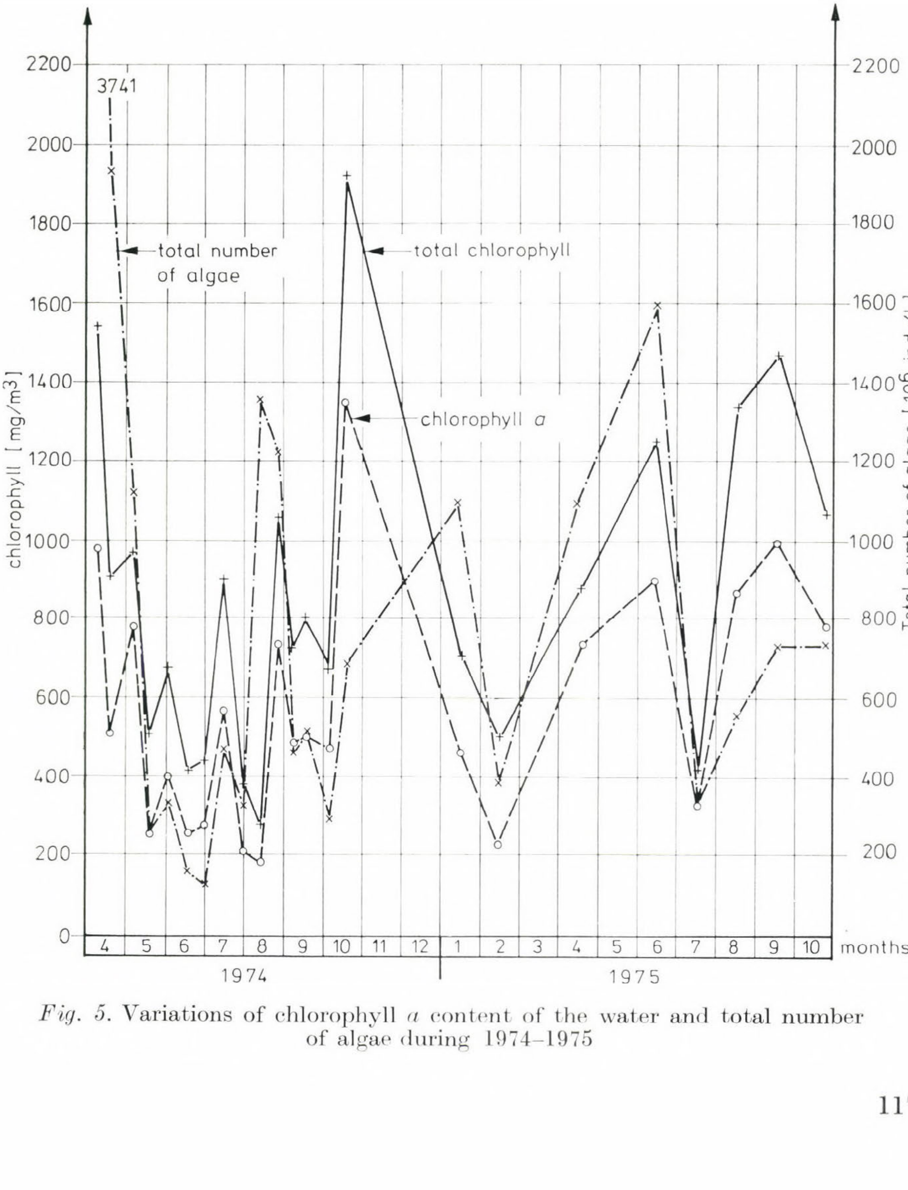

Fig. 5. Variations of chlorophyll *a* content of the water and total number of algae during 1974–1975

MANAGEMENT APPROACH TO RECTIFY THE NEGATIVE EFFECTS OF EUTROPHICATION

Seeking the possibilities of how to improve the water quality of the pond, several laboratory and field experiments have been made. First, we tried to aerate the water artificially. But this method producing promising results in some deep ponds (Ohle 1973) did not work in this case. Owing to the shallowness of the water, mud was stirred to such an extent during artificial and superficial aeration that under the influence of the released nutrients the total number of algae increased by about 25–50 per cent. Another possibility was the rotation of the pond water, as a rule together with filtration and chemical treatment. Microfiltration was made with screening cloths of 20, 40 and 60 μ mesh size and chemical treatment with different amounts of ferrosulphate, ferrochloride and aluminum sulphate. In the vertical flowing clarifier the water flowed at a speed of 0.4 mm per sec, and its retention time varied between 1.5 and 2.5 h. Then the water was conducted through an 80 cm thick rapid filter filled with sand of 0.8–1.2 mm granular diameter, at a speed of 4–5 m per h. The purification process was hindered by the specific chemical character of the water and by the circumstance that a great part of the suspended material consisted of dead bacteria, algae, protozoa, rotifers, crustaceans and their remnants. These formed a mucous mass on the filter surface. The use of diatomaceous filter allowed better purification under laboratory conditions, diminishing the amount of suspended particles, the total number of algae and the chemical oxygen demand to the minimum level. However, a considerable part of dissolved nutrients could pass through this filter, too. This procedure could be considered really successful only after long-lasting recirculation through the regenerated filter. These treatments have shown that the usual water rehabilitation technologies are not suitable for purifying the water body of Pond 'Sóstó'. This is why possible ways of water exchange have been examined, first under laboratory conditions, then by an experiment in the Bathing pond. At first we tried to choose the most suitable type of water for refilling the pond. Ground-waters were not suitable due to their high nutrient content. Top-water originating from deep wells of the waterworks of Kótaj proved to be the most appropriate. The influence of the remaining bottom deposit in the pond on the quality of supplying water has also been analysed. It has been found that nutrients previously accumulated in the 10–15 cm thick mud layer promote eutrophication. When clear water taken from the waterworks of Kótaj was poured into an aquarium on a 2–3 cm thick mud layer taken from the bottom of the Bathing pond, intensive water-bloom appeared as soon as on the third day. The total number of algae exceeded even the high values of Pond 'Sóstó'.

The experimental water exchange of the pond was made as follows. An about 1.5 m wide dam was built between the Bathing and Rowing ponds. The water of the Bathing pond was drained into the Rowing pond by a high capacity pump. The surplus water was taken away into Canal Igrice. Refilling of the Bathing pond started on 21st July 1975, using the water of Kótaj conducted through two 150 mm pipes into the western part of the pond. This procedure lasted for some days when muddy water became rich in nutrients. After repeated water exchanges the bottom deposits were flushed entirely.

According to chemical and biological analyses from the beginning of water exchange up to 20th October 1975, the biological quality of water improved (the figured code was 4540, i.e. oligo-mesohalobic, meso-eutrophic, beta-mesosaprobic and nontoxic; Felföldy 1974). Despite repeated water exchange, considerable amounts of nutrients remained in the mud, consequently, the pond needs further water quality improvement.

CONCLUSIONS

1. For restoring the pond, water of low total salt content, with low COD and BOD, as well as of low nutrient content (P and N) should be used (e.g. water originating from the waterworks of Kótaj).

2. Simultaneously with the water exchange, total desludgement of the bottom is needed. This may result in significant decrease of available nutrients.

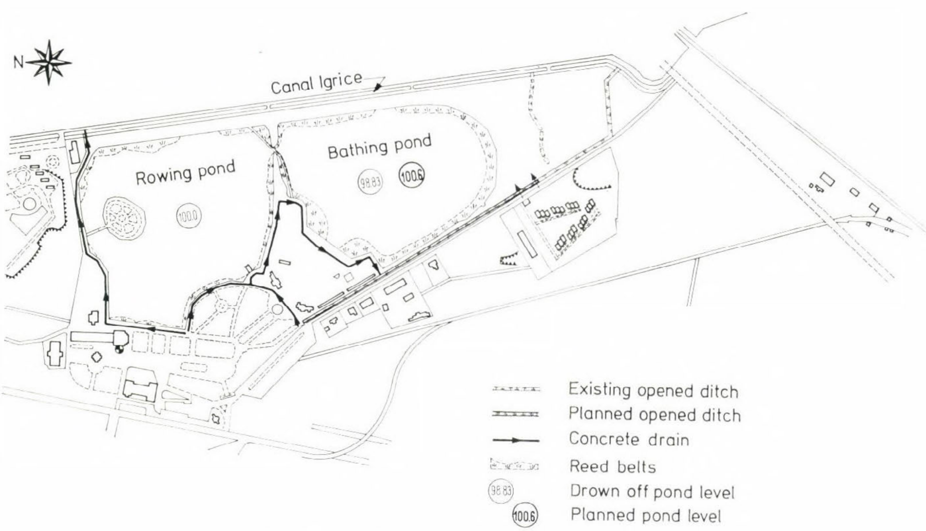

Fig. 6. Planned rearrangement of the pond

3. The direct inflow of contaminated run-off from melt and precipitation into the pond should be diminished by restructuring the shoreline. Planned rearrangement of the area is shown in Fig. 6.

4. The water supply of shallow ponds of the size of Pond 'Sóstó' should as far as possible be made independent of the surface run-off.

5. According to the results, heavy contamination by ground-water can be prevented if water level in the pond is kept 30–40 cm higher than the interstitial water.

6. Polluting effects from the neighbouring surface waters (Canal Igrice) can be avoided by maintaining the pond water on a higher level by conducting effluent sewers through a closed pipe.

7. Regular cutting and removal of reeds from the catchment area may result in a considerable decrease of nutrients stabilized in the plants.

8. Rehabilitation of Pond 'Sóstó' in the long run needs water quality control.

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MASSIVE FISH MORTALITIES CAUSED BY ALGAL BLOOMS IN EUTROPHIC ECOSYSTEMS

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Abstract

Development of noxious *Cyanophyte* blooms from high inputs of nutrients, their rapid die-off and lysis result in massive fish mortalities from oxygen depletion and high ammonia levels (summerkill). This phenomenon, representing the ultimate stage of the eutrophication process, causes seasonal large-scale catastrophes in the fisheries of many countries of the temperate and warm climatic zones. Findings of limnological research on Canadian prairie lakes over 1971–1977 are summarized with particular reference to the mechanism, ecological impact, prediction and prevention of the phenomenon.

Fish mortalities resulting from an excess growth of noxious algae and their degradation products represent the most advanced stage of lake eutrophication. This phenomenon is quite common in shallow prairie lakes of the Canadian prairie provinces. These lakes occur mostly in intensively farmed agricultural areas with self-contained watersheds. The input of nutrients from these closed basins over a long period of time, with no surface outflow from most of the lakes, brings about a steady, one-way accumulation of high quantities of plant nutrients and organic matter in the lakes. Use of fertilizer and cattle farming, typical for this area, further intensifies the natural process of eutrophication. In some of the lakes, probably the most troublesome blue-green filamentous alga, *Aphanizomenon flos-aquae*, develops (Kling 1975). Some of the *Aphanizomenon*-infested ponds experience fish kills in the summer, usually during the months of July and August, destroying partially or totally the fish stocks that were planted in the spring. The maximum concentrations of *Aphanizomenon* in these lakes are extraordinarily high. The concentrations of chlorophyll *a* reach 300–400 μg per l (the equivalent of a biomass of $>$100,000 mg per l or 2 ml per l) after settling. The prairie lakes (sometimes called potholes or sloughs) are usually small (1–20 hectares), shallow (2–5 m), thermally unstratified water bodies of moderate salinity (2000–3000 mg per l total dissolved solids, predominantly Mg^{++}, Ca^{++} and SO_4^{--} ions) (Sunde and Barica 1975, Barica 1975*a*).

From the fisheries and limnological research on the summerkill in prairie lakes it was concluded that the massive seasonal mortalities of rainbow trout (*Salmo gairdneri*) stocked each spring (Lawler et al. 1974) were caused by depletion of dissolved oxygen in lake water following the sudden death of large algal blooms and their bacterial decomposition (Barica 1973, 1975*c*). There were two major patterns of the massive die-offs so far observed in the most eutrophic lakes of the study area.

1. Midsummer collapses of *Aphanizomenon flos-aquae* blooms, taking place in July to early September;

2. Spring collapses of diatom blooms (*Cyclotella*, *Nitzschia*, *Synedra*) in combination with small greens (*Coccomyxa*, *Chlamydomonas*, *Lauterborniella*), non-*Aphanizomenon* blue greens (*Microcystis*, *Anabaena*, *Merismopedia*; May–June).

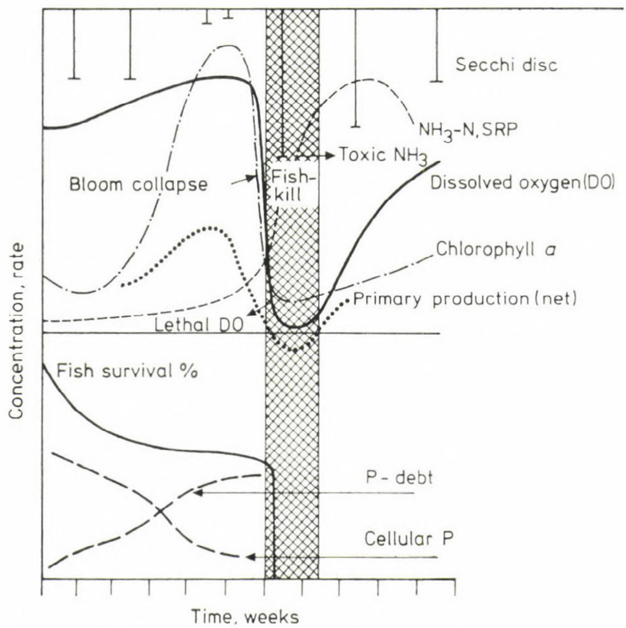

Fig. 1. Water chemistry, fisheries and algal characteristics during the course of an *Aphanizomenon flos-aquae* bloom in a summerkill lake. (Simplified after Barica 1973, 1975*c*, Ayles et al. 1976, Healey and Hendzel 1976)

Figure 1 presents a generalized pattern of the summerkill mechanism focused on parameters showing the most significant changes during and after algal collapses. The algal biomass reaches its exponential phase rapidly, with chlorophyll *a* values exceeding 100–200 μg per l and dissolved oxygen concentrations at their maximum (occasionally over 20 mg per l at the surface). The Secchi transparency is only 0.2–0.4 m in this phase, and the ammonia and the soluble active phosphate at near-zero levels. Within a few days the mass of algae starts sinking to the bottom of the lake, and the water clears downward from the surface, taking on an opalescent appearance presumably due to the release of colloidal organic matter. Simultaneously, a substantial drop of dissolved oxygen concentration is observed, accompanied by an increase in concentrations of NH_3-N (up to 2000 μg per l and over). Algae show an acute nutrient deficiency, primary production decreases sharply. This phase ends when dissolved oxygen drops to near-zero concentrations and ammonia concentration is maximal. This process is not always observed and a pattern of reversals is seen when the bloom is reduced to about 50 per cent of the original mass. In this case recovery is observed and the algae continue to grow. Dissolved oxygen does not decline below the level required by fish. This process results in a growth-decline cycle that eventually ends in catastrophic collapse of the algal bloom with the resultant anoxia described above.

The results of fisheries experiments (Ayles et al. 1974) indicated that lethal dissolved oxygen levels were the primary cause of trout mortalities following an algal collapse. Mackenthum et al. (1945) and Prescott (1948) suggested that toxic substances released by the decomposition of blue-green algae caused fish mortalities. Bioassays of aerated water from study lakes did not confirm this notion. Moreover, water temperatures were high (19–23 °C) but less than lethal levels reported in the literature. The levels of toxic un-ionized ammonia (Trussel 1972) were in some cases reached or even exceeded — this, however, happened several days after dissolved oxygen deficiency, when most of the fish were dead. It is, however, possible that the levels of algal toxins, ammonia or temperature, while not lethal, may have acted synergistically with the low oxygen levels to produce the fish mortalities.

Diurnal fluctuations in dissolved oxygen levels were insignificant in the prairie lakes because of the short summer nights in these latitudes. Low dissolved oxygen concentration persisted for several consecutive days in the whole water column.

It can be concluded that the algae cause fish mortalities indirectly: it is their sudden death and bacterial decomposition which consumes all the oxygen in the water that causes the actual fishkills.

Remedial measures to cut back the nutrient input from the lake basins and slow down the eutrophication process are not economically feasible in this area. The sources of nutrients are diffused, involving uncontrolled runoff from fertilized land and cattle farming. Summerkill risk rating and selection of lakes for fish stocking, summerkill prediction using a winter ammonia-summer chlorophyll model and/or limited use of copper sulfate to control algae were discussed as feasible alternatives (Barica 1975*b*, Whitaker et al. in press).

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HUMAN IMPACTS ON BIOMASS, POPULATION SIZE AND YIELD-PER-RECRUITS OF ASP (*ASPIUS ASPIUS* L.) IN LAKE BALATON

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Abstract

Fish populations in Lake Balaton have undergone changes which are considered to be caused by individual or combined effects of overfishing, interspecific competition from non-native species, and loadings of nutrients and pesticides. During recent years the productivity of valueless fish has increased by some 5–10 per cent in most eutrophic areas, especially in Keszthely Bay, and productivity of the more sensitive fish has declined. In addition, introduction of exotics, heavy fishing pressure, repeated summer and early spring fishkills resulted in striking changes in the composition and basic population parameters of fish. Consequently, optimal exploitation strategies for commercially important fish species should be developed in order to adjust the fishing activity to the concrete fish productivity potential of the lake.

The natural waters in Hungary, except for Lake Balaton, have not yet been studied from this aspect. However, this would be extremely important in view of the rapid environmental changes.

Little has been known about the response of fish communities in Lake Balaton to biotic and abiotic effects. The aim of this paper has been to analyse the present exploitation of asp stock by commercial fisheries in Lake Balaton and to describe its effects on biomass, population size and yield.

INTRODUCTION

The fish fauna of Lake Balaton (596 km^2) chiefly consists of cyprinids. The annual landings of fish by commercial fisheries amounted to some 24 kg ha^{-1} between 1950–1975. Changes in the fish stock of the lake induced by human interventions and cultural eutrophication (Herodek and Tamás 1975) during the last decade has directed my attention to the analysis of fishing activity (Biró 1977, 1978*a*, *b*).

Regularities in population dynamics of various fish species occurring in great numbers have already been described in detail (Biró 1970, 1971, 1972, 1973, 1975, Biró and Garádi 1974), nevertheless, there has only been some evidence on the responses to biotic and abiotic effects of species having relatively smaller virtual populations (Biró and Fűrész 1976).

Repeated fishkills in 1965 and 1975, intensive fishing, as well as introduction of exotics and eutrophication induced far-reaching changes in fish composition. The rate of these changes may be reduced by regulating fishing intensity, one of the most effective abiotic effects. Consequently, the overfishing of the stock can be avoided, or maintained at the level of optimum sustained yield.

The purpose of this paper has been to analyse responses of asp population to exploitation by commercial fisheries.

MATERIAL AND METHODS

Studies have been made using the parameters of population dynamics already described (Biró and Fűrész 1976), as well as data on catch statistics of commercial fisheries. According to the literature (Ribiánszky and Woynárovich 1962, Biró and Elek 1970, Biró 1977), data on catch statistics were collected for 1945–1975. The Balaton Fishing Company (Siófok) has placed detailed statistics at our disposal concerning the years 1964 and 1970–1975. These contained the catches of asp according to weight and number, as well as the annual fishing efforts ($\bar{f}$) expressed in hours for the five seines currently used in Lake Balaton. The given values were converted to catch per unit efforts (CPE = weight of fish taken by 100 hours of active netting).

The age-composition of the stock was assessed from the number of annuli developed on the scales of asps collected in five different areas of the lake during 1974–1975 (Biró and Fűrész 1976). Considering it more or less representative, the age-structure of the population has been estimated according to this age-distribution.

Instantaneous total mortality coefficients ($\bar{Z}$) were calculated from catch curves showing the age distribution of the stock (Ricker 1975). Linear regressions of catches according to weight (C_w) and number (C_N) against the annual fishing efforts ($\bar{f}$) were calculated. The same relationship between the instantaneous total mortality coefficients ($\bar{Z}$) obtained for different years and that of the annual number of hours spent with active fishing ($\bar{f}$) has also been estimated. Its intercept on the ordinate gives the best estimate of natural mortality (M) and the slope of this line is the 'catchability coefficient' (q). Fishing (F) and natural mortalities (M) were separated according to the formula:

$$\bar{Z} = \bar{F} + \bar{M} = q\bar{f} + M$$

(Beverton and Holt 1957, Ricker 1975). Rate of exploitation has been estimated by Cushing (1968) and Ricker (1975). Since the age-distribution of asps caught from Lake Balaton has not been representative for the whole population, and specimens older than 9+ to 10+ are very rare, the rates of mortality and exploitation have been calculated for age-groups 4+ to 10+.

Assuming that the mortality rates and growth coefficients (Biró and Fűrész 1976) are more or less constant during the whole fishable life-span of asp (from age-group 3+ to 14+), the equilibrium yield-per-recruit and biomass values of different age-groups were derived as a function of fishing mortality (F) according to Beverton and Holt's (1957) 'dynamic pool model'. Natural recruitment of the harvested population was supposed to be $R = 1$. For the yield-per-recruit estimates in different age-groups of asp, besides the parameters of von Bertalanffy's (1938, 1957) model describing the asymptotic growth in standard length ($K = 0.1518$; $L_\infty = 68.2$ cm; $t_0 = -0.63$ yr; $W_\infty = 3884$ g) (Biró and Fűrész 1976), the maximum age observed ($t_{max} = 14+$ yr), as well as average fishing (F) and natural mortalities (M) were used.

Yield-isopleths were derived from the equilibrium yield-per-recruits by linear interpolation (Wise 1972). Probable effects of alterations in fishing intensity (F) on the yield-per-recruits and biomass values proportional to catch-per-unit effort (CPE) have also been studied in various age-groups of asp.

Initial biomass (B_0) and population size (N_0) were assessed according to Leslie's method (Ricker 1975) from data on catch according to weight and number during consecutive months of 1970. This method is applicable when the asp population is fished until enough fish are removed to reduce significantly the catch-per-unit effort, the latter being considered proportional to the stock present. The annual loss or increase of stock were assessed from the biomass and population size estimates according to the rates of mortality ($\bar{Z}$, $\bar{F}$, $\bar{M}$), growth in weight (G) and production ($P/\bar{B}$).

RESULTS

Recorded Commercial Landings of Asp

In 1945–1975 the annual fish harvest from Lake Balaton varied between 433–1963 MT (7.3–33 kg ha^{-1}) and chiefly consisted of bream (*Abramis brama* L.). The ratio of noble fish varied between 9 and 19 per cent, asp having a share of 5 to 6 per cent in it (0.4–1.5 per cent of the total catch). Annual commercial landings of asp from Lake Balaton varied from 0.4 to 28.5 MT (about 0.007–0.5 kg ha^{-1}) (Fig. 1). The highest catch was recorded in 1972 and the lowest one in 1965, when due to the mass fishkill, the annual harvest dropped from 15.6 to 6.2 MT.

Fig. 1. Annual commercial catches in Lake Balaton between 1945 and 1975

Apart from the fishkill, declines in catch curve during certain periods indicate the biologically typical overfishing of the stock.

In the period of 1964–1975 the recorded commercial landings of asp at several areas of Lake Balaton were relatively constant: 2.21–2.57 MT. The ratio of asp in the noble fish catch in various parts of the lake was different: around Tihany and Siófok it approximated 8 to 10 per cent, while in the other parts, for instance in Keszthely Bay, it was merely 3 to 4 per cent. The response of asp population was very sensitive to the circumstances producing fishkill in 1965, because the numbers caught dropped to half up to 1971, especially around Siófok, Tihany and Balatonszemes when comparing the preceding years to 1964. However, at Fonyód its amount has nearly doubled, increasing also in Keszthely Bay. Annual landings at the NE-basin henceforward decreased between 1971–1975. During the six years analysed fishermen have exploited about 12.12 MT of asp annually, and the mean weight of fish was 1119 g (Table 1).

TABLE 1

Catch Statistics for Asp (Aspius aspius L.) in Lake Balaton

Year	Annual harvest of asp		Mean weight ($\bar{W}$) (g)	Annual fishing effort ($\bar{f}$) (h)*	CPE**	
	Tons	No.			kg	No.
1964	15.64	14 250	1097	2672	585	533
1971	13.34	12 199	1093	2418	552	504
1972	10.56	9944	1062	2317	456	429
1973	10.38	9024	1150	1972	527	458
1974	10.30	9367	1100	2197	469	426
1975	12.53	10 348	1211	1941	646	533
Mean	12.12	10 855	1119	2345	517	482

* For five seines.
** Total amount of asps caught during 100 h active netting.

Catch-Per-Unit Effort (CPE) and Mortality Rates

The average annual fishing effort is $\bar{f} = 2345$ hours. Catch per unit effort in 3+ to 11+ aged asps was about 517 kg or 482 specimens (Table 1) and their mean weight varied between 1.06 and 1.21 kg. From data presented in Table 1, it has been established that despite the decrease in nominal effort ($\bar{f}$), the effective one has increased.

A linear regression of the instantaneous total mortality coefficients ($\bar{Z}$) in the function of annual fishing effort ($\bar{f}$) has been calculated. The series of

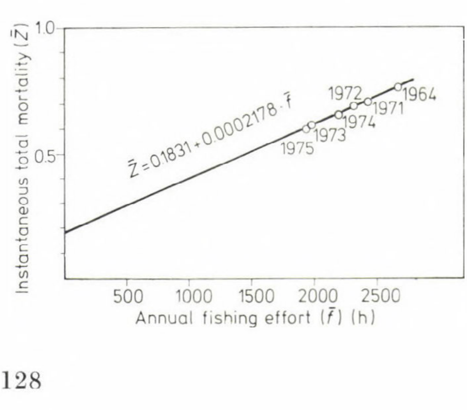

Fig. 2. Regression of the instantaneous total mortality coefficients ($\bar{Z}$) plotted against fishing efforts ($\bar{f}$) in the years 1964–1975. Its intercept on the ordinate gives the natural mortality coefficient ($M = 0.1831$)

values are placed close to the line (Fig. 2), and the S.E. of regression coefficient is very small ($s_b = 4.782 \cdot 10^{-8}$), so the regression is highly significant ($p \ll 0.001$). Mortality due to natural causes (the intercept of the regression line on the ordinate) is a constant value $M = 0.1831$. Hence, the instantaneous fishing mortality was $F = 0.4906$. Rate of survival on an average was $S = e^{-Z} = 51$ per cent, accordingly, the annual loss of the stock was $A = 1 - S = 49$ per cent. Rate of exploitation (Cushing's index) of the asp population was $E = 35.6$ per cent (Table 2). Rate of survival (S) increases

TABLE 2

Rates of Mortality, Survival and Exploitation of Asp (Aspius aspius L.) in Lake Balaton

Year	Z	F	M	S (%)	A (%)	E (%)
1964	0.7651	0.5820	0.1831	46.3	53.7	40.7
1971	0.7097	0.5266	0.1831	49.2	50.8	37.7
1972	0.6877	0.5046	0.1831	50.2	49.8	36.6
1973	0.6126	0.4295	0.1831	54.3	45.7	32.0
1974	0.6616*	0.4785	0.1831	51.7	48.3	34.9
1975	0.6058*	0.4227	0.1831	54.3	45.7	31.9
Mean	0.6737	0.4906	0.1831	51.0	49.0	35.6

Z = Instantaneous total mortality coefficient;
F = Fishing mortality coefficient; M = Natural mortality coefficient; S = Rate of survival;
A = Annual mortality rate; E = Rate of exploitation according to Cushing (1968)
* Derived from data by Biró and Fűrész (1976).

when rate of exploitation (E) decreases. When plotting the series of E and S values obtained for consecutive years their relationship can be described by a linear regression at $p \ll 0.001$ level of significance:

$$S = 0.84053 - 0.92759\,\mathrm{E}$$

Standard error of correlation coefficient is $s_b = 0.07393$. The relationship between the asps caught annually as to weight (C_w) or number (C_N) and the fishing efforts ($\bar{f}$) can also be well represented by linear regressions (Figs 3–4).

The 50 per cent length retention of the 1000 m long seines used in Lake Balaton was found for 38.7 cm and 4.3 yr of age, respectively. The selectivity of the nets basically influences the asymmetrical population structure removing the greater and older specimens from the stock.

Having calculated the regressions together, the S.E. of the regression coefficients between the selectivity indices obtained for mean standard length ($\bar{L}$) and age ($\bar{t}$), and those of instantaneous total mortality ($\bar{Z}$) and annual mortality ($\bar{A}$) coefficients, the following highly significant ($p \ll 0.001$) relationships were derived:

$$\bar{Z} = 2.5197 - 0.0477 \cdot \bar{L}$$
$$\bar{A} = 1.1761 - 0.1375 \cdot \bar{t}$$

Standard errors of correlation coefficients were $s_b = 0.077$ and 0.071, respectively.

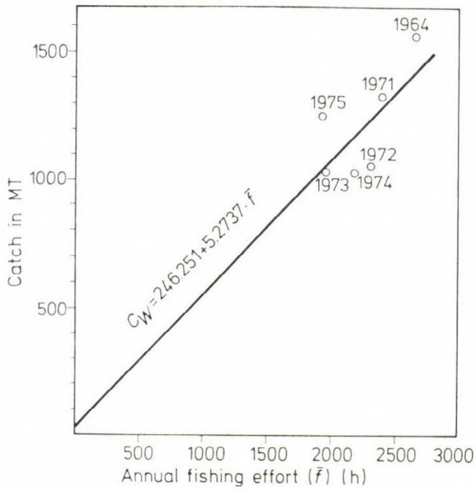

Fig. 3. Commercia landings of asp (C_w in metric tons) plotted against the fishing efforts ($\bar{f}$) of different years

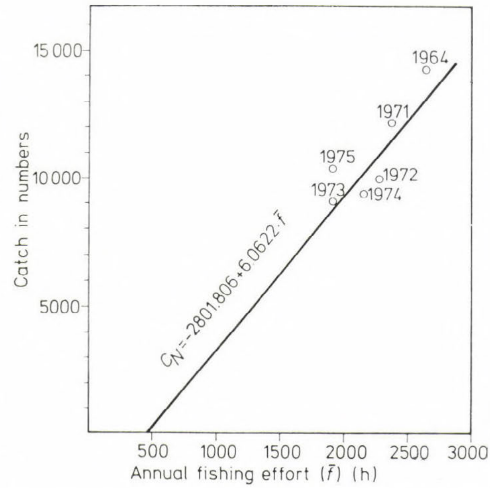

Fig. 4. Commercial landings of asp (C_N in numbers) plotted against the fishing efforts ($\bar{f}$) of different years

Biomass and Population-Size Estimates

Commercial fishery in Lake Balaton operates from melting in March until freezing in the middle of December, lasting ten months. Since it means about 988 vessel days (i.e. number of working days multiplied by 5, the number of seines used), which time, apart from closed seasons, is just evenly distributed within a month, it can be considered a single unit of fishing effort. The monthly catch divided by this time is roughly the catch-per-unit effort. Thus, one seine has been used during 19.8 days in a month. Monthly harvest of asps varied from 33 to 1966 kg, and comprised a total of 10 994 kg in 1970. Measured mean weight of 4+ to 10+ aged fish varied between 809 and 2840 g.

A linear regression between the amount of fish caught (C_w = kg) during a single unit of time (in this case it is CPE, if $t = 1$ month) and the cumulative catches (K_t) has been calculated:

$$C_t = 1340.29 - 0.04959 \cdot K_t$$

The regression coefficient (b) in this relationship is also called 'catchability coefficient'. Having determined the initial biomass of the stock from this regression, $B_0 = a/b = 27\,000$ kg (0.45 kg ha^{-1}) was obtained. Applying the constant of growth in weight ($G = 0.2314$), and the value of initial biomass (B_0) as well as the instantaneous total mortality coefficient ($\bar{Z}$), the calculated average biomass proved to be $\bar{B} = 21\,800$ kg or 0.37 kg ha^{-1}.

The total annual catch by weight in 1970 was $C_w = 10\,994$ kg (0.18 kg

ha^{-1}) and by number $C_N = 11\ 963$ specimens (0.2 ind. ha^{-1}), but the mean weight of fish caught was the lowest ($\overline{W} = 919$ g) as compared to other years. The same estimate of population size has been calculated using the number of individuals caught, and the next relationship has been obtained:

$$C_N = 1458.815 - 0.04979 \cdot K_t$$

Hence, for the initial population size we obtained $N_0 = a/b = 29\ 300$ specimens (0.49 ind.ha^{-1}). Both the limits of B_0 and N_0 are not at all symmetrical with respect to the best estimate. Average population size was derived from the $\overline{B}/\overline{W}$ ratio, that equalled with $\overline{N} = 23\ 700$ specimens (0.4 ind.ha^{-1}).

The harvestable amount of asps at the given fishing intensity can be assessed at $F\overline{B} = 10\ 700$ kg (0.18 kg ha^{-1}) consisting of about 49 per cent of the average biomass. Supposing the population to be in steady state, the total mortality is $\overline{N}Z = 16\ 000$ specimens, or $\overline{B}Z = 14\ 700$ kg (67.4 per cent). Mortality due to natural causes $M\overline{N} = 4300$ specimens, by weight it is $M\overline{B} = 4000$ kg (18.3 per cent). Mortality of 'older' fish in the stock is $\overline{N}A = 11\ 600$ specimens, nearly the same as the number of fish caught.

Under equilibrium conditions the natural recruitment to the fished stock equals with the total mortality $R = \overline{N}Z = 16\ 000$ specimens. Number of survival is nearly equal with the mortality of 'older' fish. In 1970 the estimated natural mortality of recruitment was $\overline{N}(Z-A) = 4300$ specimens.

Total annual production of age-groups 4+ to 10+ amounted to $G\overline{B} = 5053$ kg (0.08 kg ha^{-1}). This value seems to be underestimated because the annual production of the same age-groups calculated according to the $P/\overline{B}$ ratio and annual growth constants proved to be 28.6 per cent (Biró and Fűrész 1976) that would mean 6245 kg surplus production. The difference may be due to probable alterations in population structure and fluctuation in natural recruitment.

Excess weight gained during the growth processes [$(G-M)\overline{B} = 1055$ kg, or 0.018 kg ha^{-1}] compensates for the natural loss of the virtual population. On the basis of these estimates, the quantity of asp landed from the lake according to weight and number amounts to 50.3 per cent of the average biomass ($\overline{B}$) and population size ($\overline{N}$), or 40.7–40.8 per cent of their initial values (B_0 and N_0), respectively.

Yield-Per-Recruit Estimates

In Fig. 5, the equilibrium yields plotted against fishing mortality (F) are shown according to age-groups. The fishable life-span of asp is 11 years between the age-groups 3+ to 14+. Fishes older than 9+ have no economic significance due to their small ratio in the population. Yield in age-group 4+ is the highest, gradually decreasing from 5+ to 13+. Increase in equilibrium yield-per-recruits has been conversely proportional to the age of fish up to the level of $F = 0.49$. Over this, yields are hardly changeable and the curves tend to an asymptote rather than a maximum, because of high natural mortality ($M = 0.1831$). Considering the asp population to be in steady state, the yield according to the number of specimens aged from 3+

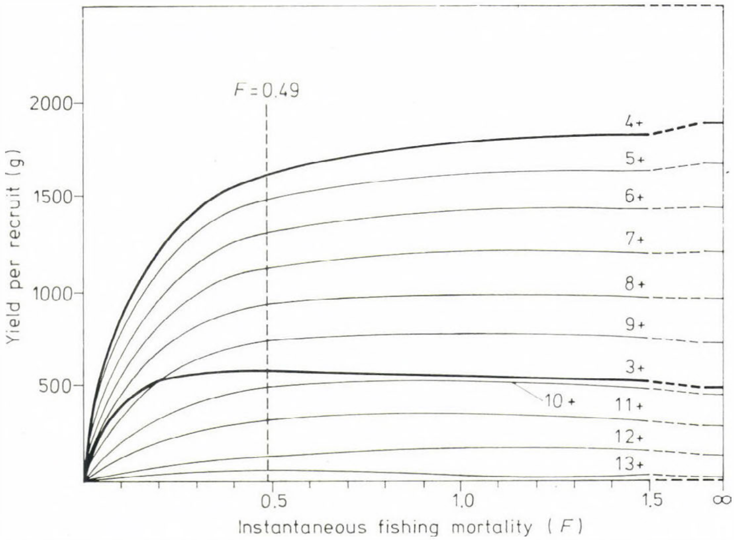

Fig. 5. Relations between equilibrium yields (in weight) and fishing mortalities (F) for various age-groups of asp in Lake Balaton, 1964–1975

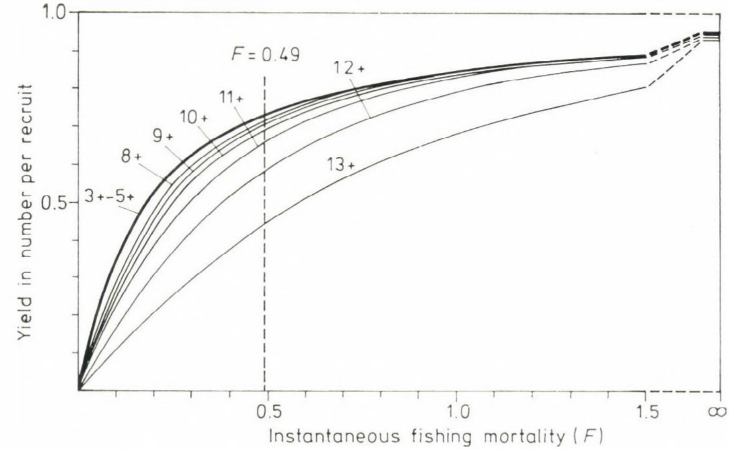

Fig. 6. Relations between equilibrium yields (in number) and fishing mortalities (F) for various age-groups of asp in Lake Balaton, 1964–1975

to 10+ does not show significant differences in the function of F. Yields in age-groups 11+ to 13+, however, are considerably different (Fig. 6).

Biomass of asp in different age-groups (proportional to CPE) varies inversely as a function of F as compared to yield-per-recruit estimates. Maximum biomass has also been shown by age-group 4+, however, it decreases to one-third of its initial value at the level of $F = 0.49$ (Fig. 7).

Population size varies along with the biomass in the function of fishing intensity (F) (Fig. 8). Catching rate of age-groups 3+ to 10+ has already been constant at $F = 0.49$.

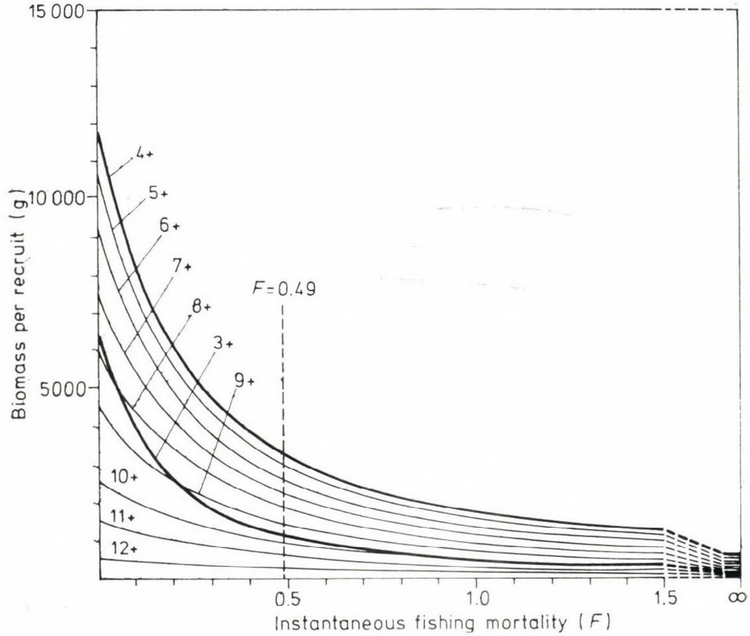

Fig. 7. Relations between biomass per recruit (assumed to be proportional to catch-per-unit effort) and fishing mortalities (F) for various age-groups of asp in Lake Balaton, 1964–1975

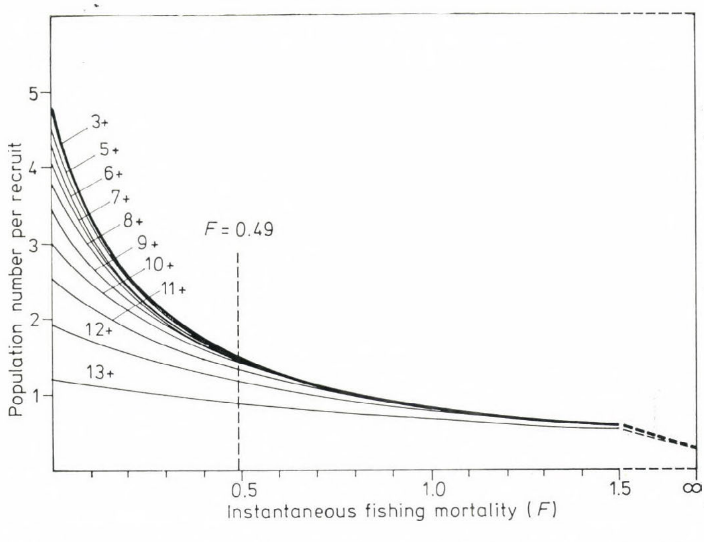

Fig. 8. Relations between population number and fishing mortalities (F) for various age-groups of asp

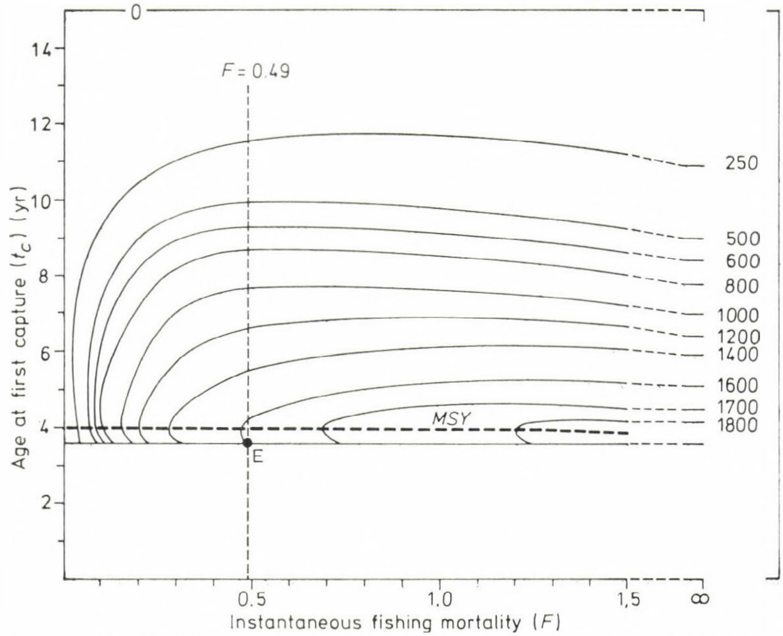

Fig. 9. Yield-isopleths of asp in Lake Balaton, 1964–1975. E = exploitation of the stock at the fixed level of $F = 0.49$; MSY = locus of maximum sustained yield

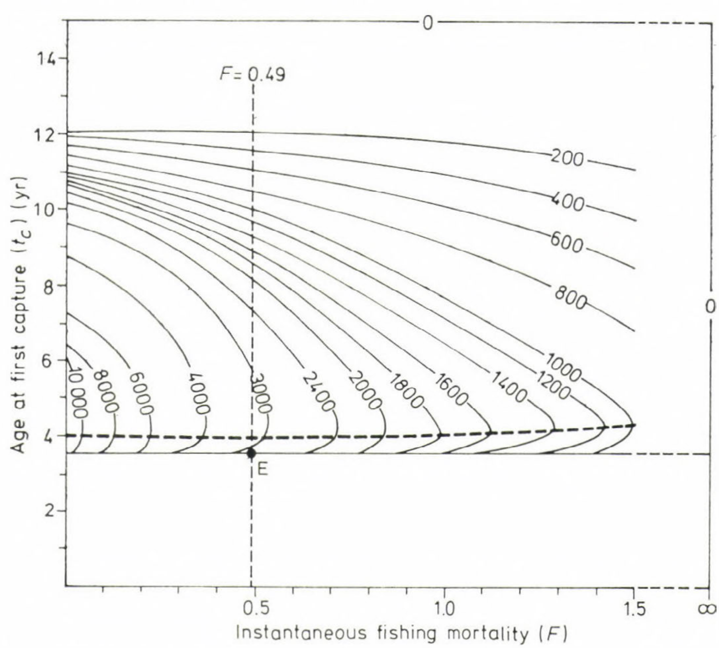

Fig. 10. Isopleth diagram for biomass of asp (assumed to be proportional to catch-per-unit effort) of exploited phase at intervals of 200 g (except from 2000 to 10 000 g). The top and right-hand borders of the diagram are the zero contour of biomass. E = exploitation of the stock at the fixed level of $F = 0.49$. Maximum sustained biomass can be achieved at $t_c = 4$ years of age at first capture (dotted line)

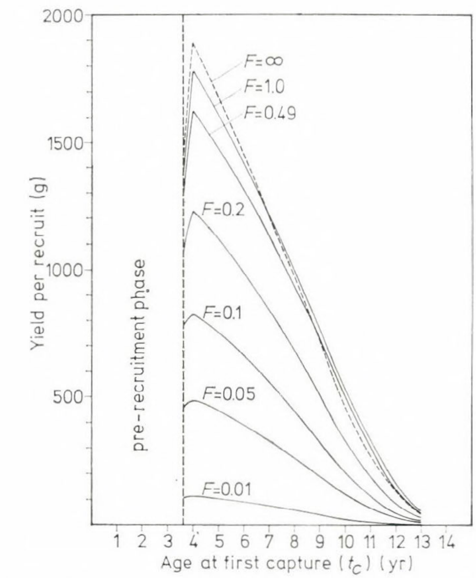

Fig. 11. Selected sections through the surfaces of yield-isopleths represented in Fig. 9

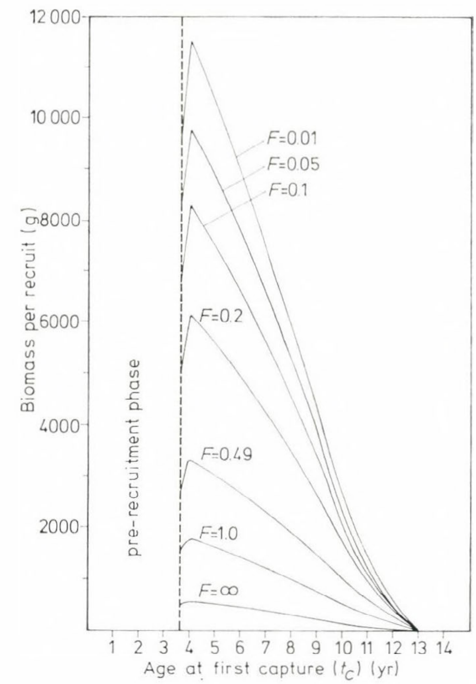

Fig. 12. Selected sections through the surfaces of biomass contours represented in Fig. 10

Point E in the yield-isopleth diagram indicates the level of exploitation of asp population. It is placed at the intersection of the level of mean age at first capture (t_c = 3.6 yr) and that of the fixed fishing intensity (F). Contrary to yield-isopleths obtained for pikeperch (*Stizostedion lucioperca* L.) and bream (*Abramis brama* L.) having dense populations in the lake, the maximum sustained yield (MSY) of asp can be achieved in age-group 4+ (Fig. 9). The locus of maximum sustained biomass is also observable in the same age-group (Fig. 10).

Selected sections through the surfaces represented by yield- and biomass contours are shown in Figs 11 and 12. They indicate that the most probable maximum yield-per-recruit at first capture (MSY) and that of the biomass (MSB) should occur in the fourth year of age of the asp, and then both values significantly decrease in its older age. 13-year-old specimens rarely occur in the stock when both yield and biomass practically equal zero. Based on the yield- and biomass contours (Figs 9–10), the optimum values can be observed between F = 0.3 to 1.0 and t_c = 3.5 to 6 years of age, respectively.

DISCUSSION

The objective of this study has been to survey what is known about the population dynamics of asp in Lake Balaton, as well as its response to human and non-human impacts.

The various factors regulating fish production are of great importance to fisheries management, but are only partially known largely owing to a lack of basic research on these problems (Swingle 1950). Fish production is known to vary widely on account ofdifferences in basic fertility of the various parts of the lake (Herodek and Tamás 1975). In addition, annual production may vary over a period of years due to differences in harvest rates, and overfishing is believed to be one of the principal causes of declines in the abundance of a species (Thompson 1950, Van Oosten 1937, 1939). The results of Swingle's (1950) experiments indicate that changes in abundance of fishes in a given body of water may often be due to shifting relationships within fish populations. Overfishing is sometimes defined from an ecological viewpoint, i.e. overfishing occurs when a stock has been reduced to such a level that it cannot readily regain its former abundance in the ecosystem. In an economic sense, overfishing occurs when a decrease in fishing effort, i.e. costs, will result in an increase in the total value of the catch in the long run (Regier 1966).

The catches of asp in various parts of the lake show that the rate of exploitation for the last three fishing seasons analysed (1970 and 1972–1973 through 1974–1975) was quite similar, and commercial landings seem to have stabilized during the last couple of years. Fluctuation in population size may be due to the uneven distribution of the stock in relationship with the differences in water quality of the lake basins (Biró and Fűrész 1976). However, there are few direct evidences to implicate the primary causes for most of these declines.

Catch-per-unit efforts and average mortality rates were also somewhat similar during the fishing season. Fishing mortality was probably high during low stock abundance, and vice versa. According to Backiel's findings, fishing mortality and total mortality of the catchable stock is inversely dependent on its density. Assuming a relatively constant population with a depensatory fishing mortality there must be a compensatory survival especially during the prerecruit phase (Backiel 1966). It is also proved by the relationship calculated between the rates of survival (S) and exploitation (E) of asp. However, the populations of most species are unstable (Regier 1966).

The environment, particularly in the most productive shallow areas, has been modified by the effect of waste disposal, runoff from fertile agricultural lands, changes in shore structure of the lake, and other factors. The changed environmental conditions in the area of asp spawning grounds (especially in Keszthely Bay and in the lower section of River Zala) and nursery areas probably play an important part in population control during the prerecruit phase.

The relationship between mortality and stock density is also affected by competition and predation (Backiel 1966). Competition may play a somewhat greater role since fry and juveniles of asp inhabit environment abundant in various fishes.

Based on our biomass, population size and yield-per-recruit estimates, the asp stock in Lake Balaton can be considered to be intensively exploited: the lower is its abundance the greater the exploitation rate at even constant fishing effort. In context of data obtained for population dynamics of various fish species inhabiting Lake Balaton, it has been established that the stability and resilience of the small asp stock is much less than that of the pikeperch and bream densely inhabiting the lake (Biró 1977, 1978*a*, *b*). The heavier is the fishing pressure the lower is its survival rate. Mortality estimates are the key to one of the most pressing problems on the lake today, i.e. estimation of rates of exploitation that will maximize long-term returns of the fishery. Fish harvest should be adjusted to the concrete productive potential of the fish populations of Lake Balaton with a view to rapid environmental changes basically influencing the population dynamics. At present unfortunately there is no available explicit theory or useful model for rational exploitation of fishery resources of seas and inland waters. The concept of maximum sustained yield (MSY) has not been sufficient for the management of inland fisheries, even its applicability has been objected by some authors (Larkin 1977). The emphasis in research should be focused on those areas where facts are needed to produce quantitative models to describe the interactions of the fish and the fishery (Regier 1966). A good descriptive model for the relationship between the eutrophication processes as measured in terms of primary production and exploitation of fish populations should be developed for the shallow Lake Balaton.

SUMMARY

Responses of the relatively small asp (*Aspius aspius* L.) population to exploitation by commercial fisheries in Lake Balaton have been analysed. Yield of asp in Lake Balaton varied between 0.007 and 0.5 kg ha^{-1}. The quantity of asp caught annually at different areas of the lake is sufficiently constant (2.2–2.57 MT).

The catch-per-unit effort (CPE) varied between 456 and 646 kg yr^{-1}, the rate of exploitation of the stock proved to be about 35.6 per cent, but the rate of production ($P/\bar{B}$) was only 28.6 per cent. Annual loss of the population is about 49 per cent and the rate of survival is 51 per cent. The selectivity of the seines used in Lake Balaton basically influences the asymmetrical population structure removing the greater and older specimens of the stock. Asp in Lake Balaton has been intensively exploited.

Rates of survival and exploitation heavily affect the size and stability of the harvestable asp population especially after the prerecruit phase. Rate of exploitation varies inversely in the function of stock abundance even at constant fishing effort. Under constant fishing pressure the stability and resilience of the relatively small asp population have been decreasing. These alterations are due to shifts in age-structure of the population and fluctuations in natural recruitment. These changes can be reversed to some extent by reducing the fishing intensity and enlarging the current mesh size of seines.

Assessed initial biomass (B_0) of the stock was 27 000 kg (0.45 kg ha^{-1}) and the average biomass ($\bar{B}$) proved to be 21 800 kg (0.37 kg ha^{-1}). Accordingly the initial population size (N_0) was 29 300 specimens (0.49 ind.ha^{-1}) and the average population size ($\bar{N}$) was 23 700 specimens (0.4 ind.ha^{-1}). The quantity of asp landed from the lake according to weight and number amounts to 50.3 per cent of the average biomass and population size or 40.7–40.8 per cent of their initial values, respectively. Under equilibrium conditions the natural recruitment to the fished stock equals the total mortality (16 000 ind.). Excess weight gained during the growth processes compensates for the natural loss of the virtual population.

Contrary to yield-isopleths obtained for pikeperch (*Stizostedion lucioperca* L.) and bream (*Abramis brama* L.) having dense populations in the lake, the maximum sustained yield (MSY) of asp can be achieved in age-group 4+. Based on yield- and biomass contours, the optimum values can be observed between $F = 0.3$ to 1.0 and $t_c = 3.5$ to 6 years of age, respectively.

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THE PROCESS OF FISH FAUNA RESTORATION IN THE RIVER PEK FOLLOWING THE CATASTROPHIC OUTBREAK OF GANGUE FROM THE COPPER MINES 'MAJDANPEK' IN YUGOSLAVIA

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Abstract

As a consequence of repeated tectonic disturbances, a breach of the ore gangue of a copper mine occurred in Serbia in January 1974.

The ore gangue was discharged together with water into the River Pek (Danube area) destroying its entire ichthyofauna.

Studies conducted for four years showed the process of river restoration and the gradual recovery of fishes immigrating from the unpolluted upper river sections, its tributaries and the River Danube. Living conditions have been studied following the catastrophe observing the growth of fishes as well as the pollution of the river by saprobity indicators.

Waste waters from the flotation plant of the copper mines 'Majdanpek' in Yugoslavia empty into a natural depression named Valja Fundata where precipitation of effluents occurs. In January 1974, following some tectonic disturbances, a breach was formed within the limestone massif of Valja Fundata and some 10 million m^3 of the ore gangue and water discharged through the breach into the River Pek. The wave of this huge, mushy mass caused heavy damages along some 80 km of the river course between the inflow of waste water and the estuary of the River Pek. Sudden slowing down of the river flow was caused by sedimentation of the gangue along the river bank some 500 m upstream from the outbreak point at Valja Fundata. The gangue was extensively deposited also along the right bank of the River Danube (near Veliko Gradište).

The gangue wave covered extensive areas of arable land with sterile deposits, thus threatening water sources and causing breakup of the flotation for a certain period. Moreover, the wave destroyed almost the entire biota in the River Pek, particularly its fish fauna.

Losses were caused to waterpower engineering, agriculture, ore mining and industry. In order to define and estimate the damage to the River Pek ecosystem, as well as to establish the rate of self-purification of the river and its sanitary conditions, studies were made by the Institute of Ore Mining, the Institute for Waterpower Engineering 'Jaroslav Cerni' and the Institute for Biological Research 'Sinisa Stankovič' in Belgrade. The study was extended to hydrological, meteorological, morphological and hydrodynamic indices, chemical and mineralogical characteristics of the river water and its sediments, as well as qualitative and quantitative composition of the biota. The study also involved the process of restoration of fish populations, and the effects of pollutants in the catchment area of the River Pek.

METHODS

The River Pek was studied for four years after the gangue outbreak (1974–1977) during periods of high and low water level. The qualitative and quantitative composition of the fish fauna was studied parallel to other components within the same sections. Collecting was made by electrofishing. The length of sections fished amounted to 200 m (Fig. 1).

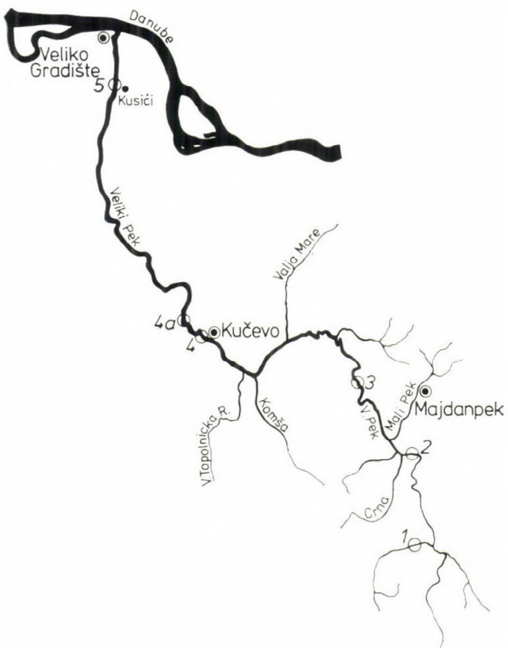

Fig. 1. The River Pek from its source to its estuary (1 : 200 000). Sampling points: *1*. Jasikovo: unpolluted river course. *2*. Just below the deposition site (outbreak point). *3*. Accumulated waste waters from the town of Majdanpek and the flotation plant of the copper mine. *4*. Kučevo: Waste waters from the town and from timber industry. *4a*. Kučevo limekiln industry. *5*. Village of Kusići: estuary of the River Pek

The fish fauna was studied in the upper, unpolluted course of the river as well as along 80 km of the river course having been subjected to the brunt of the catastrophic discharge of the ore gangue and also permanently exposed to urban, industrial, agricultural and ore mining pollution (section 5). The growth of fish in the River Pek was studied after the catastrophe in comparison with other rivers in Serbia. The growth analyses were carried out according to Monastyrsky's method (1926), using the scales located immediately below the dorsal fin. Heavy metal contamination of fishes was studied according to the methods described by Cairns et al. (1975), and Malacea (1968).

RESULTS AND DISCUSSION

The River Pek takes its source in the Homoljske Planine mountains, at an altitude of about 1065 m a.s.l. Its upper course assumes the character of a mountain stream. Its middle and lower courses flow partly through lowlands. The River Pek receives along its course many clear, unpolluted mountain tributaries. The river empties into the Danube below Veliko Gradiste (see Fig. 1).

The River Pek has for decades been receiving waste waters and other wastes from the copper mines of 'Majdanpek', from nearby factories, villages and towns causing chronic pollution along some 80 km of its course. After the catastrophic gangue outbreak in 1974, young and adult fish were killed due to a heavy load of ore gangue.

Prior to the sudden large-scale pollution, the river was inhabited by 19 fish species: 12 species of cyprinids, 3 percids, 1 *Salmo* sp., 1 *Esox* sp., 1 *Gobio* sp. and 1 *Cottus* sp. (Jankovič 1967, Jankovič et al. 1972).

The upper course of the river having the character of a mountain stream is inhabited by the following species: brown trout (*Salmo trutta m. fario* L.), bullhead (*Cottus gobio* L.), gudgeon (*Gobio gobio* L.), minnow (*Phoxinus phoxinus* L.), and barbel (*Barbus meridionalis petenyi* H.). The most common species in the middle course of the River Pek were chub (*Leuciscus cephalus* L.), nose-carp (*Chondrostoma nasus* L.) and barbel. In the lower course, strongly affected by the Danube, various cyprinids predominated. Some other species like stone loach (*Noemacheilus barbatulus* L.), pike (*Esox lucius* L.), perch (*Perca fluviatilis* L.), 'chop' (*Aspro zingel* L.) and 'little chop' (*Aspro streber* S.) immigrate into the River Pek for feeding and spawning.

Restoration of the fish fauna started in summer 1974. Six months after the gangue outbreak young specimens of chub appeared in the vicinity of the tributaries of the River Pek. During autumn, eight months after the gangue outbreak, the restoration process became more dynamic, and a more diversified fish population was noted: 'riffle minnow' (*Alburnoides bipunctatus* B.), barbel, chub, nose-carp and rudd (*Scardinius erythrophthalmus* L.). The fish populations consisted of young, but not more than three-year-old (immature or just matured) specimens.

Biochemical investigations in autumn during high water level showed the presence of sufficient amounts of soluble gases and nutrients. In all localities the water was highly saturated with oxygen, the pH being within the limits favourable for the existence of the biota. In the middle and lower stretches of the river a high percentage of suspended inorganic matter was detected. In addition, there were intolerable amounts of copper (0.36 mg per l), iron (4.4–33.3 mg per l), ammonia and arsenic acid. Increased concentrations of gangue minerals containing copper and other rare sulphide-bound metals were registered downstream in the river. The gangue minerals of minor specific weight were more easily washed away towards the Danube. An increased chalcopyrite content was also demonstrated. Harmful effects of the large quantities of gangue deposited in the river and on the banks were due also to the specific granulometric composition of the gangue (grain size < 0.2 mm).*

Analysis of the qualitative and quantitative composition of the fish fauna in 1975 showed that in the unpolluted upper part of the river, species characteristic of the upper courses of mountain rivers occurred, i.e. chub, barbel, bleak (*Alburnus alburnus* L.) and minnow.

* Results obtained by the Institute for Ore Mining, Institute of Waterpower Engineering and Institute for Biological Research.

The river segment immediately below the outbreak point of Valja Fundata is still exposed to pollution by minor quantities of ore gangue. In 1976 it was established that restoration of the fish fauna was in progress in that most endangered part of the river. Anyway, in 1977, because of continued activities to stop further deposition no fish could be caught in that part.

In the polluted part of the river receiving waste waters from the town of Majdanpek and from the mines, the fish population has increased in comparison with the previous year. The predominant species were 'riffle minnow', barbel and chub. The process of restoration was faster than in the previous year suggesting that living conditions in the River Pek were improving.

Downstream of the town of Kučevo, the river is densely inhabited by cyprinids. The fish population consists mainly of chub and barbel. Mostly young, immature specimens predominate. Occurrence of wels (*Silurus glanis* L.) near the inflow point of the waste waters of a limekiln plant proves that the effects of the Danube are detectable up to the town of Kučevo.

The restoration of fish fauna seems to be the most rapid in the lower stretch of the River Pek, near its estuary, due to the immigration of fishes from the Danube (Jankovič 1967) (Table 1).

TABLE 1

Composition of Fish Fauna in the River Pek (1975)

Sections	Composition of fish fauna	Occurrence (per cent)	
		June	October
1. Upper, unpolluted river course	*Barbus m. petenyi* H.	78.6	—
	Phoxinus phoxinus L.	14.2	—
	Leuciscus cephalus L.	7.1	80
	Alburnoides bipunctatus B.	—	20
2. Just below the deposition site (outbreak point)	—	—	—
3. Accumulated waste waters from the town of Majdanpek and the flotation plant of the copper mine	*Alburnoides bipunctatus* B.	73.3	—
	Barbus m. petenyi H.	13.3	—
	Gobio gobio L.	6.7	—
	Misgurnus fossilis L.	6.7	—
	Leuciscus cephalus L.	—	100
4. Below the town of Kučevo (waste waters from the town and from timber industry)	*Leuciscus cephalus* L.	78.5	100
	Barbus m. petenyi H.	21.5	—
4*a* Below the town of Kučevo (limekiln industry)	*Leuciscus cephalus* L.	100	90
	Silurus glanis L.	—	10
5. Estuary of the River Pek (Village of Kusići)	*Leuciscus cephalus* L.	33.3	—
	Perca fluviatilis L.	33.3	—
	Rhodeus sericeus amarus L.	33.3	—

In 1976, i.e. in the third year of the river's restoration, more abundant fish populations were established as compared with the previous years. Spawning conditions as well as the conditions of growth and diet were more favourable. The ore gangue had been considerably washed away from the river banks and the bottom. It remained only in the slow moving parts and in the river arms. Bottom fauna (Trichoptera, Ephemeroptera, Gammaridae) was also being intensively restored.

In June 1976, in the clear, unpolluted mountain part of the river, the most abundant species was chub (57.1 per cent), followed by barbel (28.5 per cent) and minnow (14.4 per cent). In the same month of the previous year, barbel was the predominant species, making up 78.6 per cent of the total catch. Hence this species is decreasing in number due to uncontrolled sport fishing.

At present, the most common species at the breach point of the depony is barbel, constituting 83.3 per cent of the total catch, followed by chub (12.5 per cent) and 'riffle minnow' (4.1 per cent). Occurrence of these fish species indicates a rapid restoration of the most affected river part. In the areas of ore mines, urban and industrial waste waters, exclusively barbel was found (100 per cent). Immediately under Kučevo, too, barbel was the only species (100 per cent). In the lower stretch of the river fish populations are particularly abundant and diversified. Barbel amounts to 75.5 per cent of the total catch, while chub to 17.7 per cent, 'riffle minnow' to 4.4 per cent and roach (*Rutilus rutilus* L.) to 2.2 per cent. This points to increased immigration of Danubian fishes into the lower stretch of the River Pek.

In order to accelerate and facilitate the restoration of the fish fauna in the River Pek, fish stocking was done in April 1976, planting one- to two-year-old fish, namely carp (*Cyprinus carpio* L.) and tench (*Tinca tinca* L.). Carp originated from the River Sitnica and tench from a fish pond. The stocking was done using 400 kg of carp and 400 kg of tench. This had no significant effect since no fishing restrictions were applied.

In 1976, water was highly saturated with oxygen. The pH and the amount of nutrients were favourable, especially in the lower river course. Close to the point of discharge of the mine waste waters there were high levels of suspended matter, whereas at the point of urban waste water discharge, high quantities of organic matter were found. In the river sediments, beside nonmetallic minerals, quartz, feldspar, lisane and clay, there were various quantities of metallic minerals: magnetite, limonite, titanium oxides (data from analyses of the Institute for Ore Mining and Institute for Biological Research).

Since fish growth is a good indicator of the pollution level of a particular water course, the growth rate was studied in the indigenous fish species inhabiting the river most densely after the gangue outbreak.

The population of chub consisted mainly of young fish (from 0+ to 3+ years). The 2-year-old specimens were the most numerous. Analysis of growth in length and weight has shown that chub in the River Pek achieves a moderate growth rate. In the first year it reaches 7.9 cm and 10.8 g, in the second year, 12.5 cm and 31.6 g, while in the third year, 18.5 cm and 119.0 g. The average growth in length and that of in weight proved to be more intensive between the second and third years than between the first and second years.

Comparison of the growth rates of this species in the rivers Pek, Timok and Radovanska (a tributary of the Crni Timok) and in Lake Bor (Danube Basin) (Jankovič 1970) has shown that it is nearly the same in the first year, with the rate decreasing to some extent in the River Pek in the following years (Fig. 2). The growth in weight of chub in the River Pek is variable. In the first year it is rapid, then follows a retarded growth in weight as compared with the specimens from other streams, whereas in the fourth

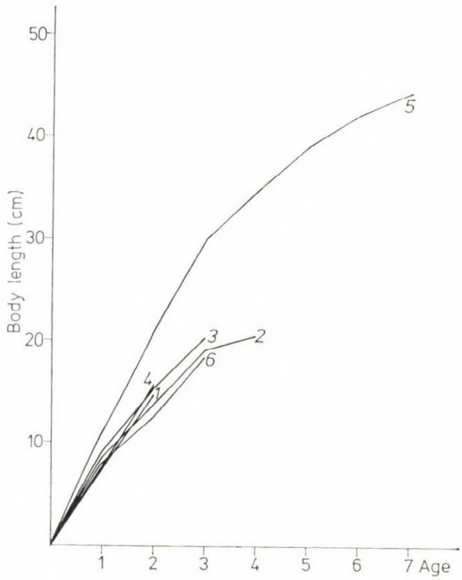

Fig. 2. The growth in length of chub (*Leuciscus cephalus* L.) from some rivers and lakes in Serbia. *1.* Svrlj. Timok; *2.* Trg. Timok; *3.* Crni Timok; *4.* Stream Radovanska; *5.* Lake Bor; *6.* River Pek

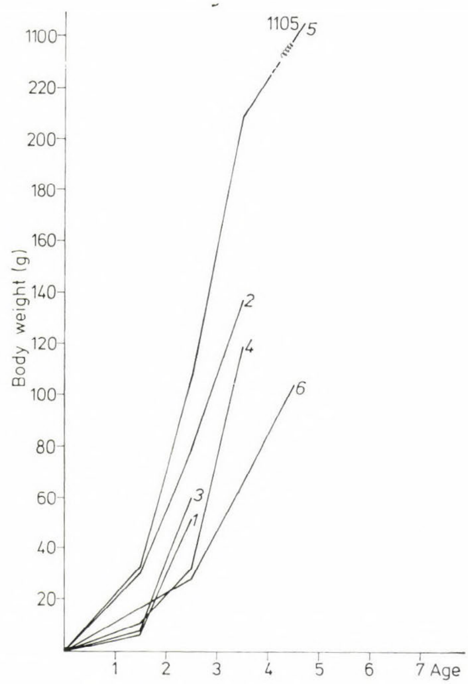

Fig. 3. Weight increase of chub (*Leuciscus cephalus* L.) from some rivers and lakes in Serbia. *1.* Svrlj. Timok; *2.* Crni Timok; *3.* Stream Radovanska; *4.* River Pek; *5.* Lake Bor; *6.* Trg. Timok

year (3+) it exhibits an average value. Chub from the River Crni Timok and the Lake of Bor grows very fast in the first and second years, the lake specimens growing later incomparably faster than the others (Fig. 3).

The growth of chub in the River Pek reflects relatively favourable feeding conditions due to rapid restoration of algae (Cladophora and Diatomea), insects and young fish constituting its primary food during the first years after the gangue pollution of the river.

The growth of barbel, another abundant species after the gangue pollution of the River Pek, was studied in 0+ to 4+-year-old specimens, the only ones that could be found in the river. In the first year the specimens reach 5.9 cm, in the second year 9.4 cm, in the third year 13.0 cm and in the fourth (only one specimen examined) 18.7 cm. Analysis of the growth in weight shows that the specimens reach only 7 g in the first year, 8 g in the second, 18.5 g in the third year and 86 g in the fifth (4+).

Comparing the growth of barbel in the River Pek and other mentioned rivers of the Danube Basin, it was concluded that the specimens from the River Pek show stunted growth in length. The specimens from Crni Timok and the Lake of Bor grow incomparably faster (Fig. 4).

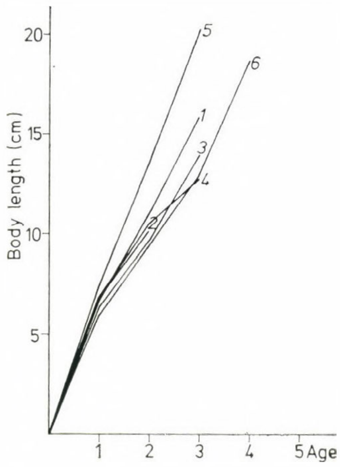

Fig. 4. The growth in length of barbel (*Barbus meridionalis petenyi* H.) from some rivers and lakes in Serbia. *1.* Crni Timok; *2.* Svrlj. Timok; *3.* Trg. Timok; *4.* Stream Radovanska; *5.* Lake Bor; *6.* River Pek

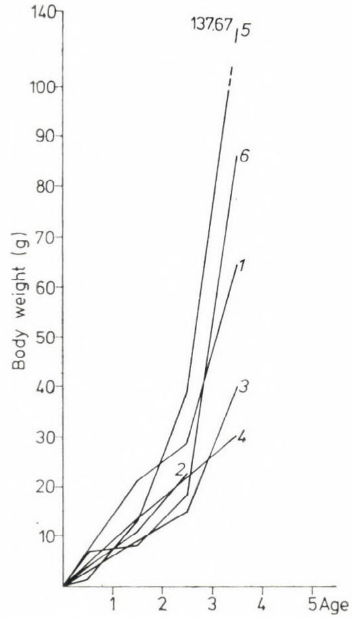

Fig. 5. Weight increase of barbel (*Barbus meridionalis petenyi* H.) from some rivers and lakes in Serbia. *1.* Crni Timok; *2.* Svrlj. Timok; *3.* Trg. Timok; *4.* Stream Radovanska; *5.* Lake Bor; *6.* River Pek

Analysis of the weight increase shows that in the first and the fourth year barbel grows faster, and specimens from Lake Bor grow very fast (Fig. 5).

Larvae of chironomids predominated in the diet of barbel. Trichoptera, some larvae of Baetidae or Heptagenia (Ephemeroptera) and some gammarids were also found in the gut content of fish.

It should be stressed that the hydrological régime of the River Pek was most favourable in 1974 and 1975, and these years were regarded as water-abundant years (mean annual flow at the station Kučevo amounted to 6.98 m^3 per sec in 1974). It provided conditions for a relatively rapid washing away of the precipitated gangue from the river bottom and banks by a large water mass. The remaining gangue has consolidated and has been covered by a sand and mud layer enabling restoration of the biota.

After the ore gangue pollution, the fauna of the River Pek consists of Ephemeroptera, Trichoptera, Plecoptera, Diptera, Coleoptera and other species available as food for fish. The diet of chub, barbel and 'riffle minnow' consists primarily of benthic elements and periphytic algae. Consumption of various chironomids was very high suggesting the rapid restoration of the bottom fauna. In the River Pek, similarly to other rivers exposed to permanent pollution, living conditions are changeable affecting the occurrence of particular fish species as well as their feeding and spawning conditions. The growth of the examined fish species under the present conditions points to a relatively satisfactory situation after the catastrophic pollution. A gradual improvement will occur in the following years if adequate protection measures and management are ensured.

Attempting to define the pollution level of the River Pek after the gangue outbreak according to the presence of particular species as indicators of saprobity, it can be established that six months after the catastrophe the first colonizing species was chub, an indicator of beta-mesosaprobity. Eight months after the catastrophe a single specimen of 'riffle minnow', an indicator of oligosaprobity appeared, followed by rudd (beta-mesosaprobic) and nose-carp (beta-alpha-mesosaprobic). Throughout the river course also barbel, a species mainly characteristic of unpolluted clear waters, occurs (Jankovič 1973).

In 1975, in the second year after the catastrophe, the population was already more diversified. In the upper part of the river, fish species known as oligosaprobic indicators were recorded, whereas the rest of the river had gradually been invaded by fish species known as indicators of beta-mesosaprobity (chub, perch, wels) or of beta-alpha-mesosaprobity [gudgeon, bitterling (*Rhodeus sericeus amarus* B.)]. 'Riffle minnow' is found near the mouth of the tributaries. Dominance of barbel in the river part affected by the pollution suggests improving conditions in the River Pek.

Two regions of the river are distinguished at present: the oligosaprobic regions including head waters and the upper river course, and the middle and lower course exposed to permanent pollution representing at present, four years after the gangue pollution, a beta-alpha-mesosaprobic region.

In view of the increased metal contents in the water of the River Pek, the most abundant fish species were analysed in respect to lead, copper, zinc and nickel content (Table 2). All the examined species contained increased amounts of zinc, whereas gudgeon and barbel contained increased amounts of copper.

TABLE 2

Heavy Metal Contents (mg per kg) in Fishes from the River Pek

Heavy metals	*Barbus m. petenyi* H.	*Leuciscus cephalus* L.	*Gobio gobio* L.
Pb	0.5	0.1	0.5
Cu	5.4	1.5	8.5
Zn	52.0	15.0	22.5
Ni	0.3	0.1	0.1

SUMMARY

In January 1974, following some tectonic disturbances, a breach was formed in the depony of the copper mines of 'Majdanpek' in Serbia. The discharged ore gangue got into the River Pek and killed the entire fish fauna over about 80 km of the river.

Restoration of the fish fauna started six months after the catastrophe by the appearance of young specimens of chub (*Leuciscus cephalus* L.). During 1975 and 1976, recolonization was intensified so that at present, 4 years after the accident, 6 out of 12 species of cyprinids, 1 out of 3 species of percids and 1 *Cobitis* sp. occur in the affected part of the river, while esocids are still lacking. The fish populations are not numerous. Wels (*Silurus glanis* L.) occurred in small numbers in the River Pek as a new species immigrating from the Danube.

Restoration proceeds by immigration of the fishes from the upper, unpolluted part of the river, from the tributaries and from the Danube.

The growth rate of the most abundant fish species chub and barbel (*Barbus meridionalis petenyi* H.) suggests relatively favourable food conditions in the River Pek after restoration of the bottom fauna.

Increased heavy metal contents have been established in the body of the fish species: zinc was found in barbel, gudgeon (*Gobio gobio* L.) and chub, and copper was found in gudgeon and barbel.

According to the presence of identified fish species indicating saprobity, at present two regions are distinguished in the River Pek: (*i*) the oligosaprobic region including head waters and the upper river course not exposed to the gangue pollution, and (*ii*) the beta-alpha-mesosaprobic region, including the middle and lower river courses affected by the catastrophic gangue pollution and permanently exposed to industrial, urban and ore mining pollution.

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THE ROLE OF FILTRATOR MOLLUSCS RICH IN CAROTENOID IN THE SELF-CLEANING OF FRESH WATERS

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Abstract

Large fresh-water bivalvian molluscs (*Unio, Anodonta*) are able to filter 20–40 l of polluted water in 24 h and to extract nutrient compounds from the filtered water. They can concentrate inedible organic and mineral residues as a detritus, which is degraded at the next stage of the aquatic biological system of self-cleaning. Molluscs play a fundamental part in the self-cleaning system.

Tolerance of the molluscs of low oxygen levels and the action of toxic agents and their ability to survive in anabiotic state under unfavourable conditions are important for the functioning of molluscs as a component of the biological self-cleaning system in polluted waters.

Molluscs with high carotenoid content are strongly resistant to environmental pollution. The population of these molluscs increases in response to pollution. On the other hand, the population of molluscs with low carotenoid content decreases when pollution increases.

Carotenoids together with haemoproteins and some respiratory enzymes form a special intracellular organoid (carotenoxysome) which is apparently capable of producing energy under conditions when mitochondria are not efficient. Carotenoids provide for an oxygen reserve in the carotenoxysome (by acting as accumulator of oxygen or of equivalent electron acceptors).

The preservation and cultivation of molluscs rich in carotenoid are very important aspects of positive human actions in the protection of the environment.

INTRODUCTION

At present the problems of biological self-cleaning of polluted waters are of great importance because the anthropogenic pollution of waters often exceeds permissible limits. The role of phytoplankton, zooplankton, microorganisms and higher water plants in self-cleaning is well known. But far less is known about the outstanding role of molluscs in the self-cleaning of polluted waters and their possible use in the system of biomonitoring.

THE ROLE OF MOLLUSCS IN THE CLEANING OF POLLUTED WATERS

The participation of bivalve filtrator molluscs in the cleaning of polluted waters is connected with the mode of their nutrition. The large bivalves, e.g. *Unio* and *Anodonta* can filtrate as much as 20–40 l of water per 24 h, extracting this way suspended particles of both organic and inorganic nature, as well as a number of organic compounds (Table 1). Filtered

TABLE 1

The Rate of Water Filtration by Some Bivalve Mollusc Species in Relation to their Body (Shell) Length (ml per mm per h) at 16–17 °C

Species	Rate of water filtration	References	
Anodonta piscinalis	14.44 ± 0.70	Kondratiev	1963
Unio tumidus	11.90 ± 0.54	Kondratiev	1963
Unio pictorum	8.36 ± 0.42	Kondratiev	1963
Dreissena polymorpha	4.80 ± 0.80	Kondratiev	1963
Mytilus edulis	25.00 ± 60.00	Voskresensky	1948
Mytilus edulis	26.25 ± 32.72	Willemsen	1952
Cardium edule	15.48	Willemsen	1952

microorganisms and particles of organic nature enter the digestive system of the filtrator molluscs, while 'inedible' particles (including mineral oil drops and other pollutants) are ejected through the delivery syphon in the form of clods. This waste contains organic compounds together with wrapping slime and serves as a complex food for microorganisms. Microorganisms, in turn, serve as food for detrivorous animals, among them apparently some gastropod molluscs.

The efficiency of self-cleaning of polluted waters depends mainly on the quantity and activity of the filtrator molluscs because the ability of microorganisms to mineralize the organic waste and mineral oil depends on the preliminary concentration of these substances by molluscs (Fig. 1).

A number of experimental data allows us to judge the efficiency of the cleaning activity of the fresh-water molluscs. For example, 200 *Unio* having the average weight of about 70 g, were used for cleaning the water of the River Don which was polluted by anthropogenic waste in a quantity of 50 mg

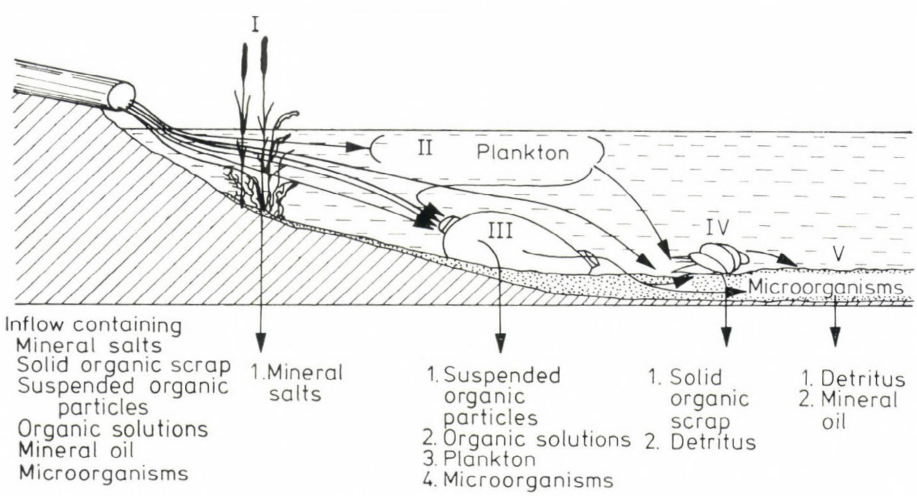

Fig. 1. The interrelation of higher water plants (I), phyto- and zooplankton (II), microphagous filtrator molluscs (III), detrivorous molluscs (IV) and microorganisms (V) in the biological self-cleaning of natural aquatic systems

per l of suspended particles. The molluscs filtered 3 m³ water in 24 h and after that the quantity of suspended particles was 0.2 mg per l (Skadowsky 1961). This means that one mollusc removed 99.5 per cent of the particles from 15 l water each day.

Naturally, in more polluted waters the rate of cleaning decreases. The molluscs *Unio* and *Anodonta* kept in the water of the Irtish-Karaganda Canal filtered 5 m³ of water per 24 h. The quantity of suspended particles decreased from 562 mg per l to 238 mg per l, while the amount of dissolved organic compounds decreased from 395 mg per l to 254 mg per l (Bervald). Thus the molluscs removed 58 per cent of the suspended particles and 35 per cent of dissolved organic compounds, i.e. 1.5 kg of suspended particles and 0.7 kg of dissolved organic compounds per day. A total amount of 63 kg dry weight of organic matter was precipitated in a container in 27 days. During the experiment the average weight of one mollusc increased from 219 to 245 g, i.e. by 26 g. The whole weight of the molluscs in the container increased by almost 3 kg.

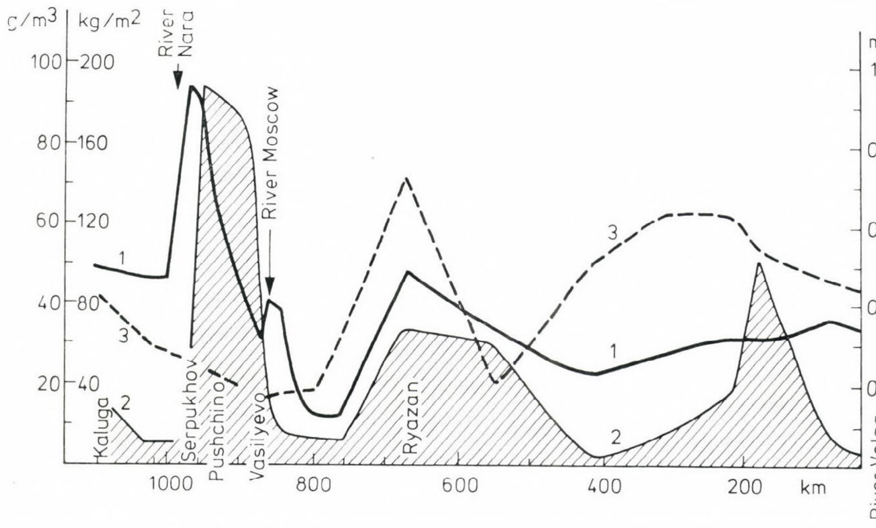

Fig. 2. The quantities of pollutants, in g per m³ (curve 1), of the molluscan biomass, in kg per m² of river-bed cross-section (curve 2), and the rate of water flow, in m per sec (curve 3) along the River Oka in 1959 (after Shadin 1964)

The role of molluscs in the cleaning of polluted natural waters can be seen from the investigations carried out on the River Oka (Shadin 1964*a*). Near Kaluga, the number of filtrator molluscs is not very large and the pollution of the water (50 g of suspended particles per 1 m³) along a distance of about 80 km remains practically the same (Fig. 2, curve 1). Near the town of Serpukhov, the pollution of the River Oka sharply increases, reaching 95 g of suspended particles per 1 m³. Eighty km below Serpukhov, the density of large filtrator molluscs (*Unio* and *Anodonta*) reaches 150 specimens per m² of bottom, and water pollution decreases by 70 per cent; the level is about 30 g of suspended particles per 1 m³ of water at the inflow of the River Moscow into the River Oka. Thus, owing to the high density of filtrator molluscs (Fig. 2, curve 2), the River Oka is cleaned of industrial and other wastes entering the river at Serpukhov at a distance of 50 km. Unfortunately, the same cannot be stated about the River Oka below the town of Kaluga, because the number of filtrator molluscs is not large enough there.

Comparing the results of investigations carried out on the River Oka in 1923–1924 and 1959 (Shadin 1964*b*) it can be seen that the level of pollution had increased in the course of 35 years, especially below Serpukhov. Together with an increase in pollution at some places in the river, a sharp increase in the number of the filtrator molluscs *Unio* and *Anodonta* was recorded. For example, below Serpukhov in the region of the town of Pushchino, the biomass of molluscs increased from 0.75 kg to 183 kg per 1 m^2 of river-bed cross-section, which is an almost 200-fold increase. The density of *Unio* and *Anodonta* in some places reached 158 specimens per m^2 of bottom. The density of the gastropods *Viviparus viviparus* also increased to 594 specimens per m^2 (Table 2). It should be noted that increased pollution

TABLE 2

Population of Molluscan Species (in Number of Specimens per m^2 of Bottom at Two Stations on the River Oka in 1959) (Shadin 1964)

Species	Pushchino	Vasilyevo
Unio pictorum	60	26
Unio tumidus	72	46
Anodonta piscinalis	26	—
Anodonta anatina	—	24
Viviparus viviparus	594	—
Sphaerium rivicola	54	—

causes a considerable growth of bivalves like *Unio* and *Anodonta*. Other species such as *Sphaerium solidum* and *Pisidium supinum* become unstable with the increase of water pollution, and their number declines sharply. In some places of the river they have entirely disappeared (Shadin 1964*b*).

THE PHYSIOLOGICAL MECHANISMS UNDERLYING THE TOLERANCE OF MOLLUSCS TO ENVIRONMENTAL POLLUTION

It can be seen that only those filtrator molluscs are capable of cleaning water which are resistant to pollution, including the increase of toxic agents, mineral salts and a decrease in the oxygen content of the environment.

Among the physiological mechanisms providing for the high resistance of molluscs to unfavourable environmental conditions one should be particularly emphasized: this is the ability of the molluscs to fall into anabiosis when the rate of metabolism in the cells sharply declines and, consequently, the oxygen uptake from the environment also decreases. Under these conditions, bivalves close their valves tightly, stop their filtrating activity, and there is a decrease in oxygen consumption (Salánki 1965, 1968).

It is apparently anabiosis that enables molluscs to survive for a long time in hypoxic conditions (sometimes for several days). Five specimens of *Mytilus galloprovincialis* were placed in a hermetic vessel (one litre volume). It was shown (Karnaukhov et al. 1977*a*) that when the oxygen concentration was decreased, the rate of oxygen consumption of the molluscs also decreased. After 10 h, the oxygen consumption rate was 1/10, while after 30–40 h, 1/100 of the initial value. Under these conditions, the oxygen consumption rate was 0.05 ml O_2 per h per 100 g tissue. Data in Fig. 3 show that the molluscs placed in the hermetic vessel fell into anabiosis within 6–10 h. Their oxygen consumption rate was slowed down when about 20 per cent of the initial oxygen level was still present in the vessel. In the anabiotic state the molluscs consumed this oxygen slowly and died after 7 days, when oxygen concentration became zero.

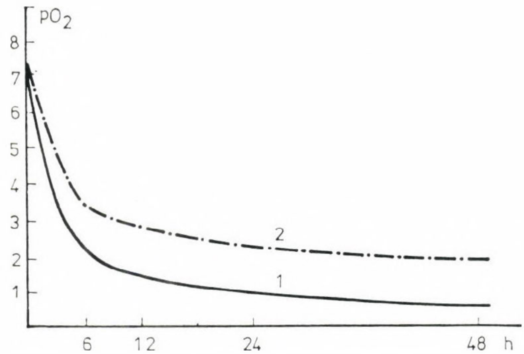

Fig. 3. Change of oxygen concentration in hermetic vessels (1 l volume) containing 5 specimens each of *Mytilus galloprovincialis* in pure sea water (curve 1) and in water polluted by 1 ml of mineral oil (curve 2)

If 1 ml of mineral oil is added into the hermetic vessel, molluscs fall into anabiosis more quickly, at an oxygen content of 40 per cent in the vessel. In this case molluscs can survive up to 12 days. It is interesting to note that *Mytilus galloprovincialis* can survive in oxygen-free water only for 8 hours (Zs.-Nagy and Ermini 1972).

The anabiotic state of the molluscs is analogous to mammalian hibernation. It is characterized by a decrease in all metabolic processes, including the synthetic activity of cells and the rate of ATP consumption in molluscan tissues. Apparently, this was the cause of the high ATP concentration measured in molluscan tissues (Zs.-Nagy and Ermini 1972) under hypoxic conditions.

THE ROLE OF CAROTENOIDS IN OXIDATIVE METABOLISM. CAROTENOXYSOME

It is evident from the physiological mechanism of the resistance of the animals to low oxygen levels and to toxic agents that some molecular mechanisms must exist that provide energy for cells, first of all for nerve cells, in hypoxic conditions. Carotenoids are considered to be part of such a molecular mechanism (Karnaukhov 1969*a*, 1973*a*).

Microspectral (Karnaukhov et al. 1966, Karnaukhov 1971) and chemical (Karnaukhov et al. 1970, Andreyev et al. 1971) investigations of the giant neurones of the mollusc *Lymnaea stagnalis* have allowed us to establish that the yellow and orange colour of the giant neurones (Fig. 4) is caused by carotenoids and haemoproteins localized in cytoplasmatic granules (Fig. 5) with specific ultrastructural organization (Karnaukhov and Varton 1971). These granules have been called 'cytosomes' by Nolte et al. (1965) who found some respiratory enzymes in them.

The absorption spectra of these granules (cytosomes) show certain changes in the neurones of molluscs in the anaerobiotic state as well as in response to inhibitors of oxidative metabolism (Karnaukhov 1968, 1969*b*, 1971) suggesting that carotenoids participate in the oxidative metabolism. Based on these findings, we have put forward a hypothesis on the function of carotenoid-containing granules in the cells (Karnaukhov 1969*a*, 1970, 1971, 1973*b*). According to this hypothesis, carotenoids may act as electron acceptors, and, together with haemoproteins, form a system of intracellular oxygen reserve (accumulator) in the cytosomes (Fig. 6). Thus, cytosomes can provide energy for the cell when the rate of oxygen penetration into the tissue is low.

The electron-acceptor and electron-donor properties of the conjugated double-bond chain of carotenoids (Pullman and Pullman 1963) allow it to connect an oxygen molecule in place of a (central) unsaturated double bond with the help of a haemoprotein. The decrease of double bonds in the conjugated chain of carotenoid leads to the loss of its colour. The colourless, oxygenated carotenoid may serve as an electron-acceptor equivalent of molecular oxygen and can be considered analogous with the oxidized form

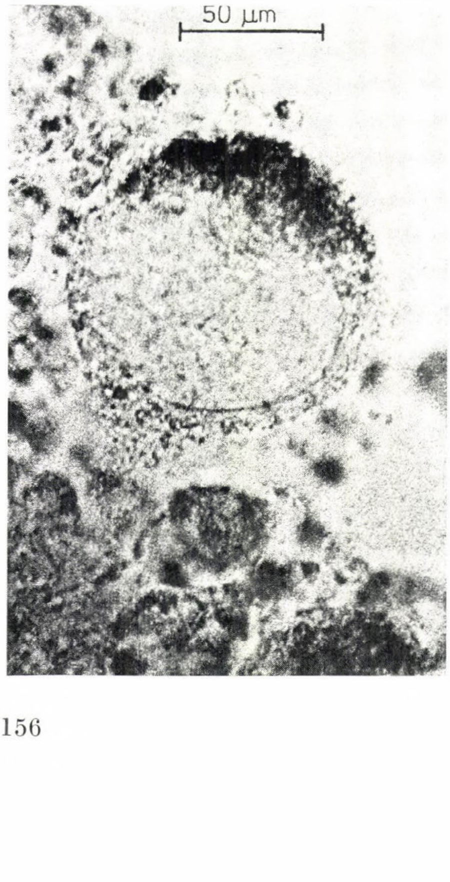

Fig. 4. Neurones of *Lymnaea stagnalis*

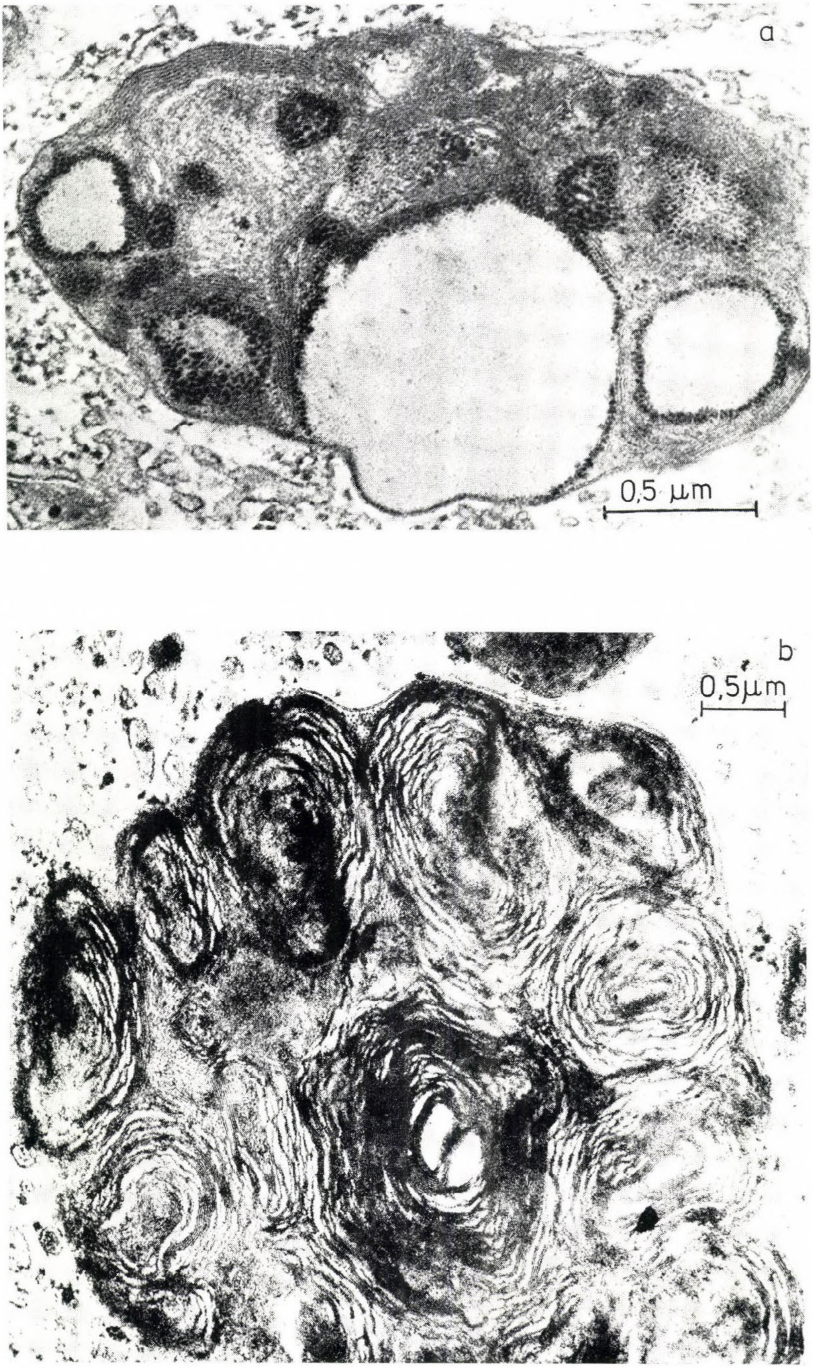

Fig. 5 a and *b*. Ultrastructural organization of carotenoxysomes (cytosomes) in *Lymnaea stagnalis* neurones of the well-known cytochrome oxidase ($a + a_3$) of the mitochondrial respiratory chain. Carotenoids can accumulate (or concentrate) oxygen (or electron-acceptor equivalents of oxygen) in resting cells under conditions of low-rate oxygen penetration (in hypoxic condition). When the cells enter an active phase under these conditions, the rate of their oxygen consumption (solid line 1 in Fig. 7*a*) becomes higher than the rate of oxygen penetration from the environment (base line 2 in Fig. 7*a*). The oxygen deficiency arising in this situation is relieved by oxygen released from the carotenoid intracellular accumulator (Fig. 7*b*) and carotenoids once more assume their colour. When the cell returns to rest, the oxygen reserve in the carotenoid intracellular accumulator can be restored (Karnaukhov 1973*a*).

Such granules rich in carotenoid (Fig. 6) are characteristic not only of the molluscan cells (cytosomes; Nolte et al. 1965, Zs.-Nagy 1967, 1971, Zs.-Nagy and Kerpel-Fronius 1971*b*, Karnaukhov and Varton 1971), but also of plant cells (carotenoid plasts; Matienko and Chabanu 1973), of the cells of warm-blooded animals and also of man (lipofuscin granules; Björkerud 1963, Karnaukhov 1973*b*, Karnaukhov et al. 1972, Karnaukhov and Fedorov 1977). Therefore they might be considered universal energy providing organoids of cells, phylogenetically older than mitochondria. Therefore we

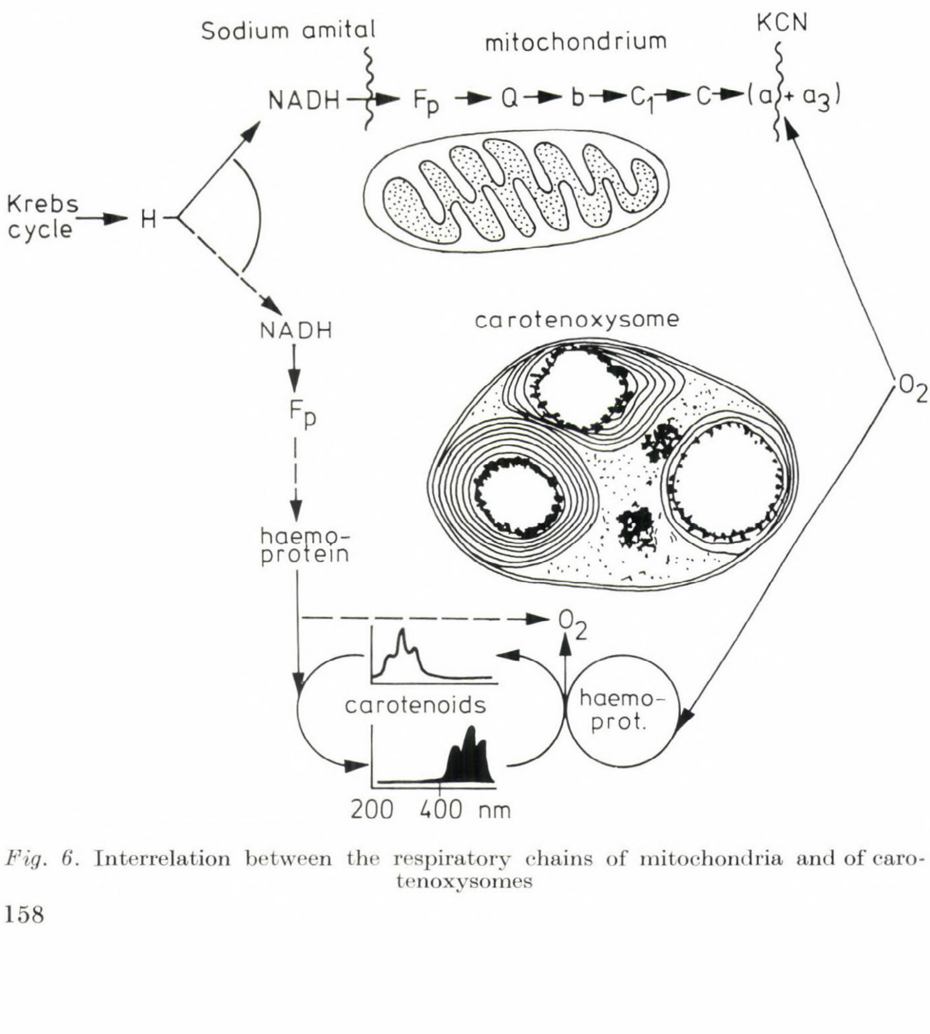

Fig. 6. Interrelation between the respiratory chains of mitochondria and of carotenoxysomes

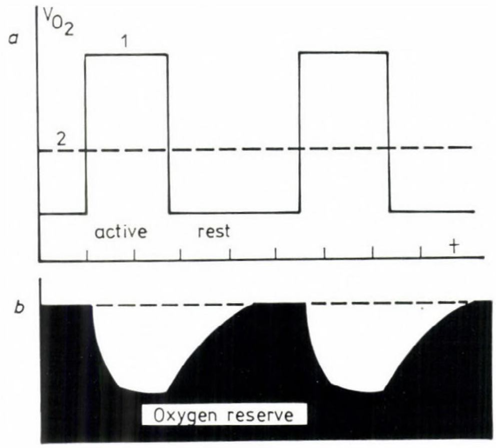

Fig. 7a. Scheme of interrelation between the rate of oxygen consumption of the cell at rest and in the active state (solid line 1) and that of oxygen penetration into the cell from media (broken line 2) under hypoxic conditions. *b.* Intracellular oxygen reserve (or its electron-acceptor equivalent) bound by carotenoid

have suggested the introduction of the term '*carotenoxysome*' for general use (Karnaukhov 1976).

The system of terminal oxidation localized in the carotenoxysomes is similar to the mitochondrial substrate–NADH system. The mitochondria are connected with carotenoxysomes by a specific regulatory mechanism whereby the system of the carotenoxysomes is not active during normal mitochondrial activity. At low oxygen levels or if mitochondrial cytochrome oxidase ($a + a_3$) is inhibited by KCN, mitochondria lose their activity, and the NADH oxidation system localized in carotenoxysomes becomes activated.

The existence of such a relationship has been proved by studying the respiration of the nerve tissue homogenate of the mollusc *Lymnaea stagnalis* (Petrunyaka et al. 1974). The homogenate contained both mitochondria and carotenoxysomes. KCN was found to inhibit the respiration of the homogenate (though it did not block respiration completely). Addition of NADH sharply increased (7–8 times) the oxygen consumption rate (Fig. 8*a*). The effects observed did not depend on the sequence of introducing the substrate (NADH) and the inhibitor (KCN). Addition of NADH to the fresh homogenate incresaed the oxygen consumption rate. This effect can be abolished by sodium amital, a specific mitochondrial inhibitor, while by adding KCN, a sudden rise in the respiration rate can be observed again (Fig. 8*b*). The degree of KCN action on NADH oxidation is dependent on KCN concentration (Fig. 8*c*).

Increase of the absorption bands of carotenoids in the absorption spectra of carotenoxysomes was found in living molluscan neurones under the influence of KCN. After treatment with sodium amital this effect was not observed (Karnaukhov 1969*b*, 1971, 1973*a*). This refers to the relationship between KCN-stimulated NADH oxidation and carotenoxysomes.

The NADH oxidizing ability of the homogenate when the respiratory chain of mitochondria is blocked by KCN testifies to the presence of a special NADH-dehydrogenase in the carotenoxysome structure (Fig. 6). Stimulation of NADH oxidation by adding KCN (Fig. 8*a*, *b*) was observed only in fresh homogenate but was absent after a day-long storage in a refrigerator at 4 °C, or when it was heated up to 90 °C. When typical mitochondrial substrates, such as α-ketoglutaric acid or succinic acid, were used, addition of KCN to the homogenate resulted in respiratory inhibition, which is quite usual for mitochondria.

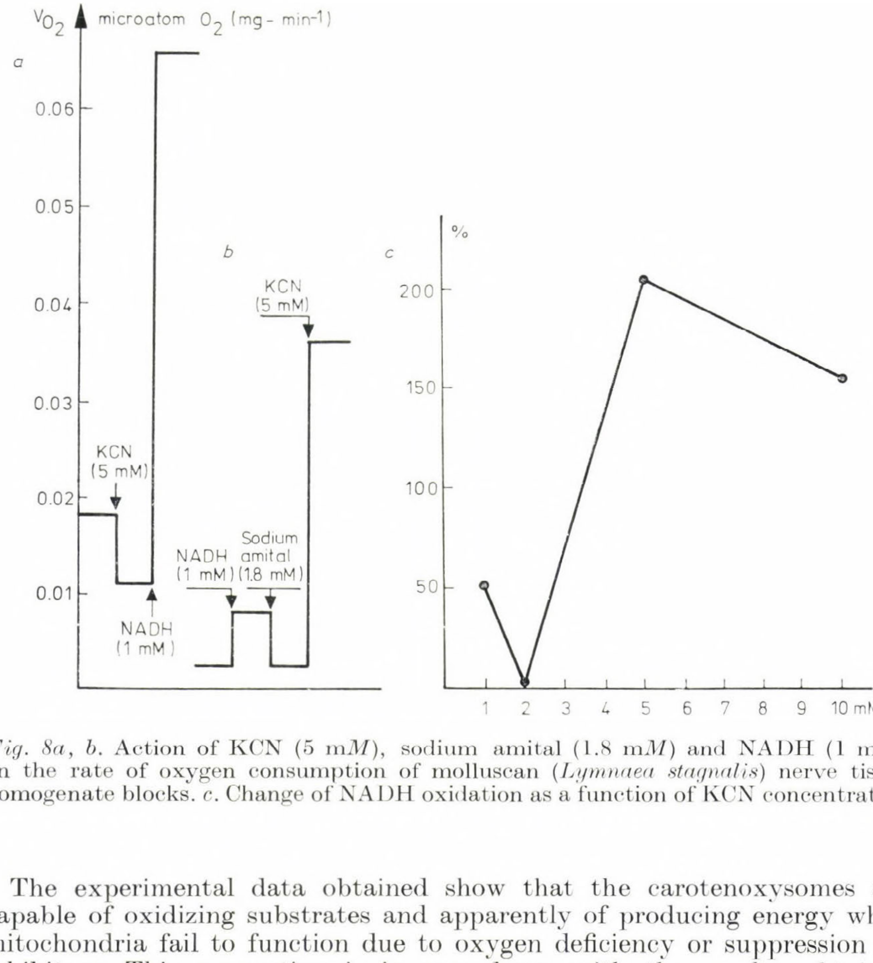

Fig. 8a, b. Action of KCN (5 m*M*), sodium amital (1.8 m*M*) and NADH (1 m*M*) on the rate of oxygen consumption of molluscan (*Lymnaea stagnalis*) nerve tissue homogenate blocks. *c*. Change of NADH oxidation as a function of KCN concentration

The experimental data obtained show that the carotenoxysomes are capable of oxidizing substrates and apparently of producing energy when mitochondria fail to function due to oxygen deficiency or suppression by inhibitors. This conception is in accordance with the results obtained for energy-dependent Sr^{++} accumulation in molluscan nerve cells (Zs.-Nagy and Kerpel-Fronius 1970*a*). It has been shown that under normal aerobic conditions, Sr^{++} is accumulated mainly in the mitochondria and, to a lesser degree, in the cytosomes (carotenoxysomes). Under anaerobic conditions the situation is reversed: Sr^{++} accumulation is observed mainly in cytosomes (carotenoxysomes), while mitochondria show very weak activity.

The above considerations allow us to suppose that the high resistance to environmental pollution is characteristic of mollusc species with high carotenoxysome content, which thus contain large amounts of carotenoids in their tissues.

Our conception was tested experimentally by studying the concentration of carotenoids in the body of several species of Black Sea molluscs with different degrees of tolerance of environmental pollution. Pollution is accompanied by a decrease of dissolved oxygen and by an increase of different respiratory inhibitors in the water (Karnaukhov et al. 1977*a*, *b*). We had access to the data on quantitative and qualitative changes of littoral animals in the Black Sea, collected over many years at the Novorossiisk Biological Station and at the Institute of South Seas Biology (Zernov 1913, Milovidova 1966*a*, *b*, 1975). The effect of oil pollution was particularly interesting from the point of view of elaborating biodegradation methods for cleaning polluted waters (Mironov 1973, Milovidova 1975).

The most important results of the study are shown in Table 3 and Fig. 9, listing different mollusc species in order of their tolerance of environmental pollution. The degree of tolerance is expressed by a system of four grades elaborated on the basis of observations on the bottom biocenosis of the Black Sea littoral (Zernov 1913, Milovidova 1966*a*, *b*, 1972).

Group I includes the pure-water inhabitants: *F. ponticus* and *D. trunculus*, which have been living recently only on the shore (sea coast). In the past these species used to live in large numbers in Sevastopol Bay and the middle part of Novorossiisk Bay (Zernov 1913). This is proved also by the large number of empty shells of these species found on the bottom of the bay. At that time the water's oxygen deficit amounted to 1.8 mg O_2 per l in the Novorossiisk Bay. The increasing pollution and silt accumulation had led to the gradual disappearance of these species from the bay; now they only live near the bay mouth.

TABLE 3

Comparison of Some Black Sea Mollusc Species as Regards Their Degrees of Tolerance of Environmental Pollution and the Carotenoid Concentration in Their Bodies

Species	S_{10}	T	N	Carotenoid concentration in mg per 100 g tissue (wet wt)	
				Total	Unsaponifiable
(1) *Flexopecten ponticus*	—	I	21	trace	trace
(2) *Donax trunculus*	—	I	28	1.32	0.33
(3) *Gouldia minima*	−10	II	46	0.01	trace
(4) *Polititapes aurea*	− 7.6	II	20	0.08	0.06
(5) *Pitar rudis*	− 6.6	II	20	0.24	0.20
(6) *Chemellea gallina*	− 2.8	II	30	0.42	0.08
(7) *Acanthocardia tuberculata*	+ 2.5	III	7	1.71	1.62
(8) *Cerastoderma glaucum*	+ 2.1	III	11	2.06	0.92
(9) *Mytilus galloprovincialis*	—	III	117	2.00	0.70
(10) *Tritia reticulata*	+ 2.0	III	571	2.05	0.90
(11) *Cerithium vulgatum*	—	III	19	2.60	1.38

$S_{10} = k(n_0/n_{-10})^k$: Population change during 10 years.
n_0 and n_{-10} Number of specimens per 1 m² of bottom recently and 10 years earlier, respectively; $k = 1$ if $\geq n_0$ n_{-10} and $k = -1$ if $n_0 < n_{-10}$.
T: Degree of tolerance of environmental pollution.
N: Number of specimens studied.

Species which populate sea areas with moderate pollution belong to group II. In 1967 the water oxygen deficit in these areas was 2.04–2.34 mg O_2 per l (Milovidova 1972). The degree of tolerance of these species, *G. minima*, *P. aurea*, *P. rudis*, and *C. gallina* (see Table 3), was established on the basis of the decrease of population from 1960 to 1969, since pollution had been increasing during this period of 10 years. The decrease in population number with an increase in pollution is characteristic of the species of group II (3–6 in Fig. 9*b*).

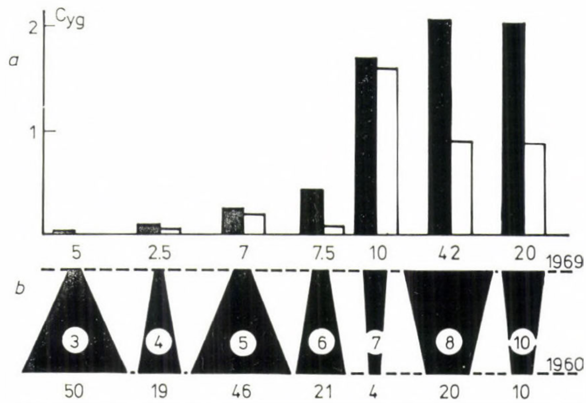

Fig. 9a. Total (black blocks) and unsaponifiable (white blocks) carotenoid concentrations (in mg per 100 g wet weight of tissue) in molluscs. *b*. Change of the population of molluscan species (No. = number of specimens per 1 m^2 of bottom) in response to pollution which increased from 1960 to 1969 in the Novorossiisk Bay. For the serial numbers indicating molluscan species see Table 3

Group III includes species inhabiting the strongly polluted area of the bays, where water oxygen deficit was 2.64–2.94 mg O_2 per l in summer 1967 (Milovidova 1972). An increase of the population parallel with increasing pollution is characteristic of the species *A. tuberculata*, *C. glaucum*, and *T. reticulata* (7, 8, 10 in Fig. 9*b*). It must be noted that the data presented in Fig. 9*b* refer to a definite region in the central part of Novorossiisk Bay, where water oxygen deficit changed from 1.9 to 3.1 mg O_2 per l during 1960–1969. Sanitary protective measures taken in the bay in 1968–1969 had led to a sudden decrease in the pollution rate. This resulted in a decrease of the water oxygen deficit and a gradual recovery of bottom biocenosis. Therefore, the data showing the molluscan populations in this transition period are not considered in the present report.

The mollusc species studied can be definitely classified into three groups on the basis of their different degrees of tolerance of environmental pollution accompanied by a decrease in dissolved oxygen concentration and by an increase in various toxic substances. Comparison of the data presented in Table 3 and Fig. 9 shows that the species of Groups II and III differ considerably as regards the carotenoid concentrations in their bodies. A low concentration of carotenoids is characteristic of group II, the population of which decreases as pollution increases. At the same time and at the same place, the species of group III are characterized by a high carotenoid concentration (0.7–1.62 mg unsaponified carotenoids per 100 g wet weight) in their bodies. These species respond to increased pollution by increasing their population.

It is believed that the existence of such a strong correlation between the degree of molluscan tolerance of environmental pollution and the carotenoid concentration in their bodies supports our hypothesis that carotenoids are involved in the adaptation of the animal to hypoxic conditions. However, the existence of species like those included in group I shows that other environmental factors, e.g. silt accumulation, may have similar effects.

It must be stressed that the specific carotenoid concentrations presented in Table 3 and Fig. 9*a* were found in animals obtained from relatively pure areas of the sea. At the same time, the species of group III (*C. glaucum*, *T. reticulata*) taken from strongly polluted areas had higher carotenoid concentrations (Fig. 10). Probably this is due to the adaptation of animals to a higher degree of environmental pollution.

Analysing the data of Table 3 and Fig. 9, it is necessary to point out that species Nos 3 to 8 belong to the order Venerida. The species *G. minima*, *P. aureus*, *P. rudis*, *C. gallina* of the family Veneridae are characterized by low carotenoid concentration and by low tolerance to environmental pollution, while *A. tuberculata* and *C. glaucum*, belonging to the same order but to the family Cardiidae, have high carotenoid concentration and high tolerance to environmental pollution as well.

According to the hypothesis formulated earlier (Karnaukhov 1969*a*, *b*, 1970, 1971, 1973*a*, *b*) the utilization of carotenoid conjugated double bonds for oxygen accumulation results in the loss of colour of carotenoid. On the contrary, the withdrawal of the accumulated oxygen from the colourless

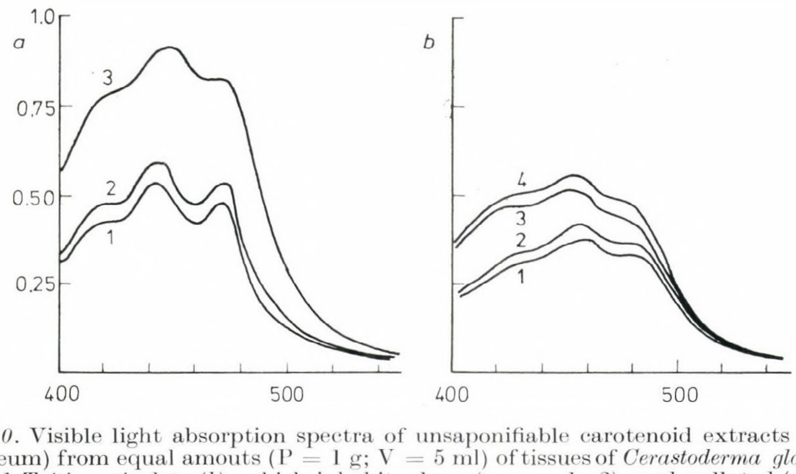

Fig. 10. Visible light absorption spectra of unsaponifiable carotenoid extracts (light petroleum) from equal amouts (P = 1 g; V = 5 ml) of tissues of *Cerastoderma glaucum* (*a*) and *Tritia reticulata* (*b*), which inhabit clean (curves 1, 2) and polluted (curves 3, 4) bays. Ordinate: optical density; abscissa: wavelength in nm

carotenoid under hypoxic conditions is accompanied by reconversion of the carotenoid into coloured form with characteristic three-band absorption spectra in the visible region.

This allows us to expect an increase in the coloured carotenoid concentration in the body of molluscs having been kept under hypoxic conditions.

To test this hypothesis, *Mytilus galloprovincialis* was studied. 250 specimens obtained from a moderately polluted area of the sea were placed in an aquarium with slow perfusion of sea water polluted by mineral oil. Under these conditions the molluscs remained in an active state (the valves were open), but they died after 50–60 h. Concentration of carotenoids in the animals was measured before, and 6, 24 and 48 h after the beginning of the experiment. The carotenoids were extracted from 20 specimens simultaneously, and the experiment was repeated four times for each point. The data obtained are presented in Fig. 11. The most remarkable increase in carotenoid concentration was detected in the first 6 h of the oil treatment and hypoxia. After 48 h carotenoid concentration reached 6–10 mg per 100 g tissue (i.e. 3–4 times higher than the initial one) and then the molluscs died within 2–3 h.

From the time of survival of *Mytilus* in oxygen-free water (8 h) (Zs.-Nagy and Ermini 1972*b*) and assuming that molluscs are in anabiotic state under these conditions (the rate of oxygen consumption is 0.05 ml per h per 100 g wet tissue), the intracellular reserve of oxygen (or its electron acceptor

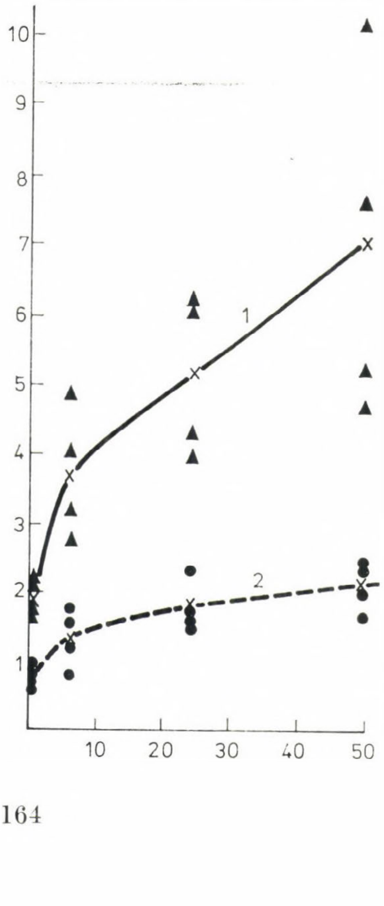

Fig. 11. Change of average concentration (in mg per 100 g wet weight) of total (curve 1) and unsaponifiable (curve 2) carotenoids in the body of *Mytilus galloprovincialis* under the action of slow perfusion (hypoxia) of mineral-oil-polluted sea water. The dots are data of the pooled tissue of 20 specimens. Abscissa: time of perfusion in h

equivalent) required for survival can be calculated. The intracellular reserve of oxygen (or its equivalent) for 8 h survival is 0.4 ml O_2 per 100 g tissue (1×10^{19} molecules O_2). Taking into account that 1 mg carotenoid contains 1.2×10^{18} molecules, and assuming that one carotenoid molecule can bind one oxygen molecule, it is clear that 8.3 mg of carotenoids per 100 g tissue is needed for *Mytilus* to survive for 8 h in oxygen-free water. This calculated value is close to the total carotenoid concentration (7.4 mg per 100 g tissue) which has been measured in molluscs kept in polluted water for 48 h (curve 1 in Fig. 11), when the intracellular oxygen reserve was not completely exhausted.

In case of fresh-water molluscs, too, it can be expected that their tolerance of environmental pollution is determined by the carotenoxysomes in their cells and, consequently, by the high carotenoid content in their tissues. The presence of the carotenoxysomes (cytosomes) in *Anodonta cygnea* L. cells was repeatedly demonstrated by investigators at the Tihany Biological Research Institute (Zs.-Nagy 1967, Zs.-Nagy and Kerpel-Fronius 1970*a*, *b*). They also showed that the carotenoid content in the tissue of the ganglia of *Anodonta cygnea* reaches up to 10 mg per 100 g wet weight (Lábos et al. 1966).

In addition, it must be remembered that carotenoids are not uniformly distributed in the animal's tissues and cells. A higher carotenoid concentration is characteristic of the nerve tissue. For example, the carotenoid concentration in the nerve tissue of the fresh-water snail *Lymnaea stagnalis* is 30–40 mg per 100 g wet tissue. Apparently, the carotenoid system for intracellular accumulation of oxygen (or of its electron acceptor equivalent) is an exclusive property of tissues and cells which are more significant for the animal's survival.

THE USE OF MOLLUSCS IN THE PROTECTION OF WATER BODIES

In conclusion, the fundamental role of the molluscs for the self-cleaning of polluted waters must be emphasized again. Among the actions directed towards the preservation of the environment the recultivation of molluscs, particularly of those rich in carotenoid in waters where they have been killed by pollution, is one of the most urgent tasks. Further, these molluscs need strong protection where they still survive; and, finally, experiments have shown that molluscs rich in carotenoid may be used as components of an artificial biocenosis with high resistance to pollution as well as for the cleaning of polluted waters.

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BEHAVIOURAL STUDIES ON MUSSELS UNDER CHANGING ENVIRONMENTAL CONDITIONS

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Abstract

The pumping behaviour of the fresh water mussel (*Anodonta cygnea* L.) was investigated under laboratory conditions with reference to artificial environmental factors, namely the presence of various pollutants.

The continuous recording of shell activity and evaluation of changes in activity proved to be good indicators for testing the effect of chemicals.

Mussels are rather sensitive to sulfhydryl blocking agents (Hg, Cd), which cause a marked shortening in their active periods. Some insecticides and herbicides shortening the mussels' activity also result in the reduction of water cleaning performed by mussels.

The visible consequences of using hazardous agents in the environment are in some cases most dramatic, like massive fishkills in rivers and lakes, severe disease or death of birds and other animals in the mainland. In water ecosystems, as a rule, massive chemical pollution originating from industry or agriculture proved to be the cause of such events. Coincidence of a number of unfavourable conditions in the environment and within the animal, e.g. low O_2 pressure, accumulation of toxic agents, sudden change of temperature, etc. may cause similarly grave damage to water life.

Although mortality very clearly shows the toxicity of various substances, in such cases only the irreversible result of environmental deterioration and the level of toxicity of various factors for a given species can be registered. However, environmental pollutants do not only cause acute mortality, but may also produce chronic diseases, influencing the physiological processes and survival of animals, and can thus result in a reduction in the number of population. For this reason, besides determining the toxic level of recently used chemicals under various environmental conditions, also investigation of the effect of sublethal doses is extremely important. This must be especially emphasized in water ecosystems because animals living there are permanently exposed to substances dissolved in the water, and also due to the possible accumulation of these chemicals in the organisms both directly from the water and via the food chain. The development of appropriate methods for demonstrating chronic, sublethal effects of hazardous agents in different species seems to be a very important task also in the biological monitoring of the occurrence of dangerous substances and situations in water ecosystems.

Among the animals inhabiting both marine and fresh waters, mussels living on the shore or at the bottom are most widespread. Mussels do not only feed at their location, but when obtaining food and oxygen they also filter the water thus taking part in the cleaning of water. The filter feeding behaviour results in the accumulation in their body of various substances present at low concentrations in the water.

Investigations on the fresh water mussel have been conducted primarily not for determining the lethal level of various agents coming from the environment, but for elucidating their influence on the behaviour of these animals.

MATERIAL AND METHODS

Experiments were made all the year round on *Anodonta cygnea* L. collected from fish-ponds and kept for several months in Lake Balaton. The wet weight of animals without shells varied between 100–120 g. The animals under investigation were kept in tanks containing water from Lake Balaton, which was exchanged regularly. Depending on the season, the temperature varied between 15 °C and 25 °C. It was not cooler in winter either, due to the fact that water was not pumped directly from the lake into the tanks, but through a water reservoir situated on top of the Institute's building.

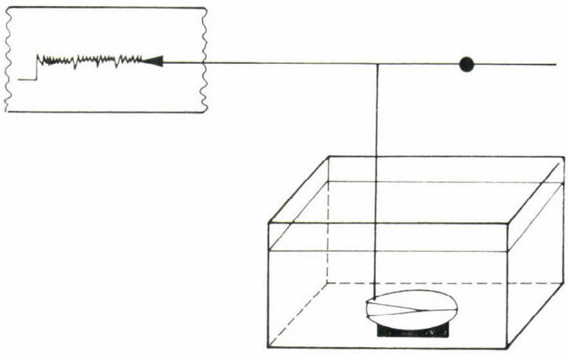

Fig. 1. Scheme of recording the activity of mussels

For monitoring the behaviour of mussels the motor activity of adductor muscles and, consequently, the pumping activity were recorded by using mussel actograph (Salánki and Balla 1964). According to this method, one of the shells is fixed, while the other is connected to a lever. The closed or open position of the shells and the contraction or relaxation of adductors can thus be recorded (Fig. 1) on a slowly rotating paper. Another method used for recording shell movements does not require the fixation of animals (Véró and Salánki 1969). This system includes electronic devices fitted to the shells and movements of the latter cause current changes fed into a recording apparatus. By using this method, the pumping activity of freely moving animals can be recorded under natural conditions. The records obtained from mussels kept under natural and under laboratory conditions were very similar, indicating that fixation of animals on one of their sides does not influence their pumping activity. Recording of activity could be continued with both methods for several weeks or months without interruption. Evaluation of data can follow afterwards.

All the experiments in which the effect of different chemicals was tested, were conducted on fixed animals under laboratory conditions. Mussels were kept separately in vessels containing 3 l water, and control activity was recorded for several days. Water was changed once daily, at a given hour. Substances previously dissolved were directly added to the vessel. After a testing period of more than 24 hours, also the test solution was changed daily.

The effect of various types of substances was investigated. Among them, heavy metals and plant protecting chemicals deserve special attention.

RESULTS

By recording the activity of *Anodonta* either in natural environment or under laboratory conditions, characteristic patterns can be obtained. In accordance with the finding of Marceau (1909) and Barnes (1955), the rhythmic pumping movement of the shells occurs as a result of fast contractions and relaxations of the adductors during the period when shells are generally in an open state. This is the active filtering period of the animal when uptake of food and oxygen occurs. From time to time this active state is interrupted by rest periods when, as a result of tonic contraction of the adductors, the shells are tightly closed for hours, communication of the animal with the surrounding stops, and food and oxygen uptake are blocked, This pattern of the filtering behaviour is called periodic activity. The duration of both the active and rest periods varies widely. It can be different even in animals living under the same conditions. Nevertheless, activity usually lasts from 10 to 20 hours, but sometimes it can be several days long,

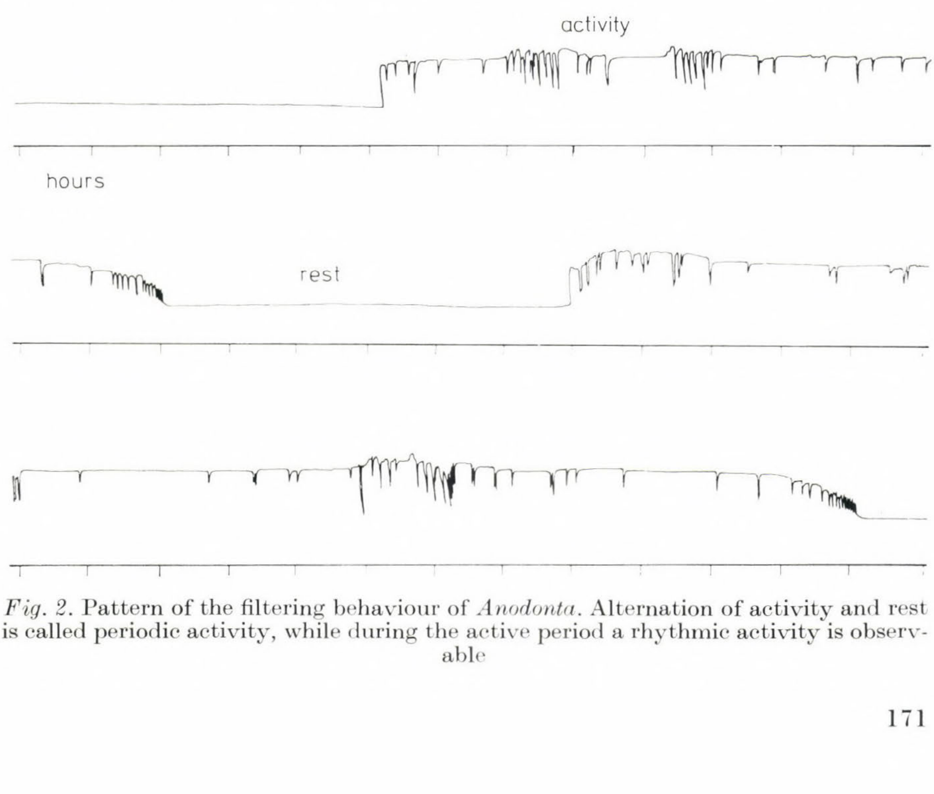

Fig. 2. Pattern of the filtering behaviour of *Anodonta*. Alternation of activity and rest is called periodic activity, while during the active period a rhythmic activity is observable

while duration of rest is normally 5–10 hours. This, however, can also be variable (Salánki et al. 1970). The frequency of rhythmic movements during the active period is variable, too. The range usually is 5–20 contractons per hour (Fig. 2).

Owing to the great variability of time and frequency values, experiments were carried out with self-control. Drugs were applied after a certain time of recording the control activity, and the effect of substances was estimated as a deviation from the pattern observed in the control period. As an indicator of the change of filtering behaviour, the mean duration of the active periods was considered.

Effect of Heavy Metals on Filtering Behaviour

Figure 3 shows a typical example of the effect of para-chloromercuribenzoate on periodic activity. With the substance added to the water, there was an obvious change in the absolute and relative length of active and rest periods. Particularly the duration of the activity shortened, and so rest periods occurred more frequently. The proportion of activity within a given time decreased to less than 50 per cent as compared to the control.

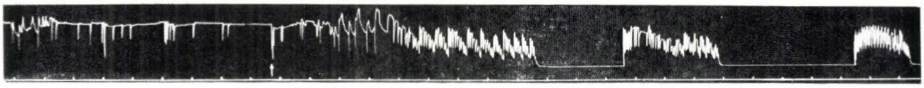

Fig. 3. Effect of *p*-chloromercuribenzoate on the filtering activity of *Anodonta*. Arrow indicates adding of pCMB (10^{-8} g per l). Time: hours

The effect produced by salts of heavy metals depended considerably on concentration, and elimination by washing out usually required several days. If concentration was too high (over 10^{-6} g per l), the animals did not survive but died as a result of intoxication.

Testing the effect of $CdCl_2$, $CuSO_4$ and $PbNO_3$, it was found that both Cd and Cu produce similar effect as Hg, but Pb proved to be ineffective on filtering behaviour. The effect caused by $CuSO_4$ — shortening of the mean duration of active periods — plotted against the concentration is

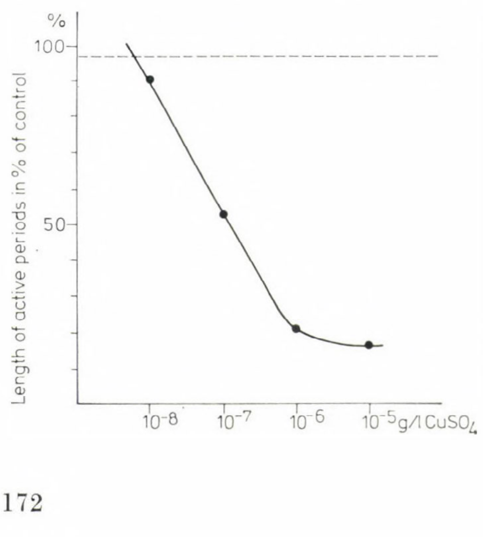

Fig. 4. Effect of $CuSO_4$ on the mean duration of active periods of *Anodonta*. Both control and treatment periods lasted for one week

presented in Fig. 4. It is remarkable that the range of $CuSO_4$ concentrations causing at least 20 per cent decrease in activity and leading to the death of animals was rather narrow, between 10^{-8} and 10^{-6} g per l. In case of Hg and Cd, this range was about the same, varying between 10^{-9} and 10^{-7} g per l. Nevertheless, these concentrations can be considered high even as compared with the levels found in polluted waters, and mussels seem to be rather insensitive to these substances in acute experiments (Salánki and Varanka 1976).

Effects of Chemicals Used in Plant Protection

Mercury and copper are not only industrial waste materials, but are also used in agriculture against fungi, being components of various inorganic and organic compounds. Besides metal-containing substances, also other pesticides and herbicides are washed into rivers and lakes greatly influencing the life of mussels. The effect of some insecticides, containing lindane, phosphamidon and phorate, has been described in another paper (Salánki and Varanka 1978). Recently, the effect of a herbicide called Gramoxon has been demonstrated. Its active compound is paraquat dichloride. Mussels did not show high sensitivity to Gramoxon, however, the range between the threshold and lethal concentrations was very narrow. Although 10^{-3} ml per l did not noticeably cut the duration of the active periods, 10^{-2} ml per l resulted already in an 85 per cent shortening of active periods (Fig. 5). Increasing the concentration up to 10^{-1} ml per l, the mussels died within 72 hours.

It is interesting to note that the death of the animals did not usually occur during the rest period but at the end of a comparatively long activity. The mussel can be considered to be dead when its adductors are totally

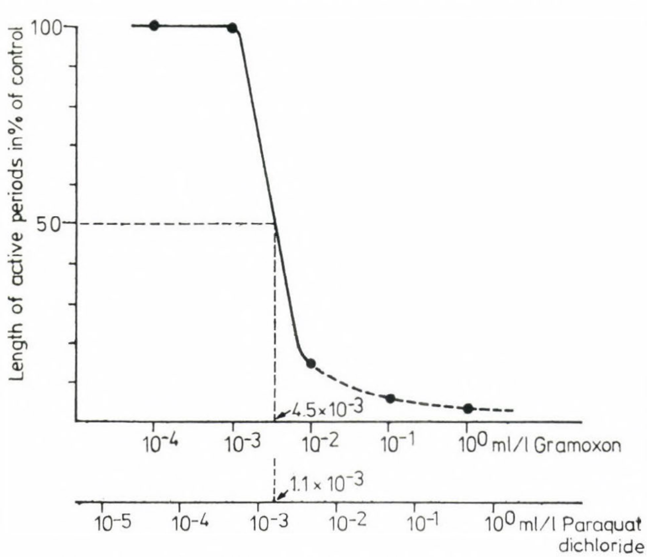

Fig. 5. Effect of Gramoxon and its active ingredient on the duration of active periods of *Anodonta*

relaxed without any rhythmic movements. Prior to this stage, as a sign of deterioration, the adductors become incapable of tonic contractions (to keep "catch"). However, for some time — sometimes even for 10 or 20 hours — they show rhythmic activity of a lower amplitude.

DISCUSSION

The functioning of adductors is necessary for all the vital processes of bivalvian molluscs. This can serve as a basis for investigating rhythmic and periodic activity being also indicative of the behaviour of the animal. Although filtration and uptake of food are mainly connected with the ciliary activity being a continuous process on the gill surface and at other parts of the mantle cavity, pumping of water with the aid of shell movements is a prerequisite for the effective work of the cilia. In previous investigations it was found that both water cleaning and oxygen uptake are concurrent with the activity of mussels (Salánki and Lukacsovics, 1967), and both of these physiological processes are depressed during the rest period. It was also shown that heart rate is lower during rest than during activity (Pécsi and Salánki 1964, Taylor 1976), and also digestion shows the same periodicity as activity and rest (Morton 1969). There is also evidence suggesting that ciliary activity itself has a different rate depending on the animal's behaviour. Of course, it remains to be seen whether pumping as a functional process influences the heart rate, digestion and ciliary activity, or whether there is a central regulation of these processes including the rhythmic and periodic activity of the adductors. As to the heart rate, it seems that the same neural mechanism which regulates the movement of the adductors is simultaneously involved in the increase or decrease of heartbeats (Pécsi and Salánki 1964). In securing the periodicity of digestion, food uptake seems to have a regulatory role (Morton 1969) and, according to Taylor (1976), heart rate is also influenced by the O_2 pressure in the mantle cavity. Food and oxygen uptake as well as excretion depend on the flow of water through the animal, which, in turn, is regulated by the pumping activity.

The substances used in our experiments influenced the pumping behaviour of *Anodonta* significantly in most of the cases. The considerable shortening of the active periods resulted in a decrease in the amount of water filtered by the animal, and consequently, oxygen and food uptake came to be depressed. Thus the rate of metabolism in the animal was probably lowered, and growth must also have been retarded. Decrease in the rate of growth (Butler et al. 1960, Frazier 1976) and also the death of various bivalves kept in waters of increased salinity or containing pollutants have been reported (Imlay 1973, Okazaki 1976).

Heavy metals are known to block the SH groups of various enzymes. In our case it is supposed that respiratory enzymes should in the first place be considered in this respect (Salánki 1960). Our earlier experiments had revealed that oxygen deficiency may cause a heavy depression in the activity of mussels (Salánki 1965), and heavy metals, especially Hg and Cd, can exert their effect via the respiratory system as well. The effect of Cu should probably be interpreted differently, and also that of plant protecting chemicals might be connected with intracellular metabolic processes.

Nevertheless, the fact that different substances, e.g. the herbicide paraquat and also insecticides like lindane, phosphamidon, phorate (Salánki and Varanka 1978) decrease the activity of mussels, shows that this effect can be regarded as a generalized reaction to the change of the chemism of the water.

In our short-term experiments, mussels proved to be rather insensitive to low concentrations of the substances used. Nevertheless, it can still be supposed that during a longer period, when accumulation of the drugs occurs, mussels would be the most suitable organisms for the biological monitoring of the quality of rivers and lakes.

Considering the various ways in which the substances used may act, probably none of them influences the physiological properties and functioning of the adductors directly. This means that all the described effects were mediated by the nervous system which can be influenced directly, but the activity and regulatory function of the ganglia can be modified also by afferent pathways arriving from peripheral and visceral receptors or by metabolites which are produced in the different organs. In the central regulation of the periodic activity of mussels and also at neuromuscular level, serotoninergic and catecholaminergic mechanisms play the key role (Hiripi and Salánki 1973). It is highly probable that all the factors changing the behaviour of *Anodonta* act through metabolic pathways or by releasing monoamines in the ganglia and in the adductors at nerve terminals. Both catecholamines and serotonin strongly influence the duration of active periods (Salánki et al. 1974), and also drugs influencing the synthesis and breakdown of monoamines change the periodicity of activity and rest effectively (Hiripi 1973).

Further investigations are required to explain both the mode of action of various substances and also the way they modify the central neural mechanism. Additional research is also needed to clarify the effect of the various types of chemicals occurring in lakes and rivers as a result of environmental pollution on the behaviour of mussels. Thereby valuable information may be obtained as to the degree to which different substances are dangerous for mussels, a species playing an important role in water purification.

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EFFECT OF SOME PESTICIDES ON THE RHYTHMIC ADDUCTOR MUSCLE ACTIVITY OF FRESH-WATER MUSSEL LARVAE

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Abstract

The toxicity of some pesticides which may enter fresh waters was examined on the larvae of fresh-water mussels. The decrease of tryptamine (TA)-induced adductor muscle activity was used as an indicator of the effect because it seemed to be suitable for the investigation of inhibitory effects. The following compounds were used: insecticides: Hungaria L-7, Dimecron-50, Thimet-10 G, Novenda, Bi-58 EC and Malathion; a nematocide: Shell-DD; herbicides: Gramoxon, Dikonirt and its active ingredient, the pure 2,4-D salt, as well as LAS, a surface-active agent.

INTRODUCTION

In the recent decade the extensive use of highly persistent pesticides and herbicides has made it necessary to examine the effect of these compounds on life in waters, as they may enter streams and stagnant waters. Their effect on adult bivalve molluscs has been summarized by Salánki (p. 169 in the present volume). However, the toxic substances may affect not only the adult specimens, but also their different developmental stages. Some information is available on the different degrees of tolerance of the mussel larvae and adult mussels (Hidu 1965, Swedmark et al. 1971). The investigations have mainly been performed on the eggs and larvae of marine species, however, concerning the balance of water ecosystems it is also important to know how the larvae living in fresh-waters can tolerate the toxicity of these compounds.

Earlier the lethal levels were generally used as indicators of the toxicity of pesticides. Wisely and Blick (1967) suggested that, due to their specialized protective reactions, the larvae of bivalve molluscs are not suitable for testing the effect of toxic substances, particularly in short-term (some hours) experiments. The shell-closing reaction caused by a strongly toxic effect reduces the penetration of the compounds into the soft parts of the larvae, resulting in considerable variations in mortality. That is the reason why recently the examination of the chronic effects has become predominant. In these experiments the changes in the development of fertilized eggs (Davis and Hidu 1969, Swedmark et al. 1971, Granmo 1972), or the decrease in the rate of larval growth (Davis 1961, Hidu 1965, Davis and Hidu 1969, Granmo 1972) was considered as the indicator of toxicity.

The present experiments were performed on the larvae (glochidia) of the fresh-water mussel *Anodonta cygnea* L. (Varanka 1977, 1978, Granmo and Varanka 1979). The larvae developing from fertilized eggs can be found in the gill-bags. The soft part of the glochidia is covered by two shell-valves hinged together with a larval adductor muscle. The adductor muscle shows two types of spontaneous movement: (*i*) rhythmic adductor muscle activity which consists of phasic contractions and relaxation; (*ii*) lasting closure resulting from tonic contraction. The nervous elements are still in an early stage of their differentiation (Herbers 1914, Zs.-Nagy and Lábos 1969), and the rhythmic activity can be interpreted as being of myogenic origin (Lábos et al. 1964*a*).

The physiological and pharmacological properties of larval adductor muscle activity of glochidia are sufficiently well known. KCl and tryptamine (TA) have a considerable excitatory effect on the rhythmic activity of adductor muscle (Lábos and Salánki 1963, Lábos et al. 1964*a*; Lábos 1966), however, TA-sensitivity decreases with maturity. In addition, the influence of some alkali metal and alkali earth-metal ions, bioactive agents as well as some other chemicals have also been examined on the rhythmic adductor muscle activity of the glochidia (Lábos et al. 1964*a*, *b*, *c*; Lábos 1967, 1970).

In our experiments the inhibition of TA-induced activity of adductor muscle caused by the examined pesticides was used as an indicator of toxicity, since the spontaneous activity generally has a very low frequency (0–0.02 cpm) thus being unsuitable for the assessment of inhibitory effects.

MATERIAL AND METHODS

The experiments were performed in winter and spring from October to May at room temperature (20–23 °C). The gill lamellae of adult freshwater mussels *Anodonta cygnea* L. containing glochidia were removed and cut into pieces, then the larvae were washed out with filtered Balaton water. The pieces of gills and the mucous material were removed with repeated washings. The larvae of each mussel were stored separately in Balaton water in a cool place at +4 to +5 °C, not longer than a week. The water was changed every second day.

Before the experiments the glochidia were divided into perspex cells, each of them containing twenty-five individuals in 0.15 ml filtered Balaton water. The cells were placed under a binocular microscope, illuminated with standard tungsten light source (6 V, 15 W) from 10 cm distance across an infrafilter type BC-17 (about some hundred Lux nominal illumination). Observations started following 30 min adaptation of the larvae to temperature and light (Lábos 1966).

Following this, the spontaneous rhythmic contractions of the adductor muscle and the number of closed glochidia in a cell were counted every minute, for 5 min. Then the Balaton water was replaced with 100 mg per l TA-solution, and the TA-induced contractions were counted for 25–30 min. The solutions of substances to be tested contained the same concentration of TA in addition to the appropriate pesticide concentration. In the next group of glochidia another pesticide concentration was tested. The effects of compounds were investigated using decreasing or increasing concentrations in every series of experiments. Thus each concentration of all pesticides was tested on 200–300 larvae originating from 4–5 adult mussels.

The effect on the spontaneous adductor muscle activity of each pesticide without TA was also tested, but we failed to observe any effect except for the high concentrations of some pesticides. The degree of toxicity of some pesticides varied with the age-dependent TA sensitivity of glochidia. Therefore, in these cases toxicity was investigated separately in a TA-sensitive group and a TA-insensitive one but the result was estimated for a mixed (in 50–50 per cent) population.

For the evaluation of the results, the total number of contractions of 100 larvae per minute (*a* per min) and the value of lasting closures in per cent of the total number of larvae ($c\%$) were calculated and graphically presented. The number of contractions of 100 larvae in 25–30 min (whole experimental period) was also calculated (Σa), and the percentage inhibition of TA-induced activity by pesticides was graphically represented as a function of the pesticide concentration. The concentration resulting in 50 per cent inhibition of TA-induced activity was graphically estimated from this curve. The solutions were freshly prepared in filtered Balaton water. The water-insoluble compounds were used as suspensions or colloid.

The following compounds were used. Insecticides: Hungaria L-7 (active ingredient 7.8 per cent lindane; made by Budapest Chemical Works, Hungary), Dimecron-50 (50 per cent phosphamidon; from active ingredient by Ciba–Geigy AG, Switzerland; made by Nitrochemical Works, Fűzfő, Hungary), Thimet-10 G (10 per cent phorate, American Cyanamid Co., U.S.A. and Werft Chemie, Austria), Novenda (25 per cent DNOC = 4,6-dinitro-*o*-cresol, Nitrochemical Industry, Fűzfő, Hungary), Bi-58 EC (36 per cent dimethoate, VEB Chemie-Kombinat, Bitterfield, G.D.R.), and Malathion (98 per cent pure; Shell Chemicals Ltd., England). Nematocide: Shell-DD (50 per cent 1,3-dichloropropene + 1,2-dichloropropane, Shell Int. Petr. Co. Ltd., England). Herbicides: Gramoxon (25 per cent paraquat-dichloride, from active ingredient by ICI Plant Prot. Div., England; made by Alkaloida, Tiszavasvári, Hungary), Dikonirt (75 per cent 2,4-D, Nitrochemical Industry, Fűzfő, Hungary) and its active ingredient, the 2,4-D-salt (93 per cent pure; Chemie Linz A.G., Austria) and a surface active agent, the linear anionic dodecylbenzene sulphonate = LAS (MODO-KEMI AB., Stenungsund, Sweden).

The pH of Balaton water was 8.45, while the pH of solutions varied between 7.9 and 8.45.

RESULTS

The insecticide Hungaria L-7 (HL-7) produced in concentrations higher than 10^{-1} g per l a slight excitation of the adductor muscle with spontaneous activity (Fig. 1). At the same time a lasting closure occurred in part of the larvae. The frequency of contractions and the percentage of closed larvae increased with higher concentrations.

Figure 2 shows the TA-induced adductor muscle activity of the TA-sensitive and TA-insensitive groups. A considerable difference in the maximum frequency of TA-induced activity as well as in the total contraction number could be observed between the two populations. In both groups 5×10^{-1} g per l concentration of HL-7 was required for the inhibition of TA-induced activity. Following this inhibition the total inhibition was different in the two populations.

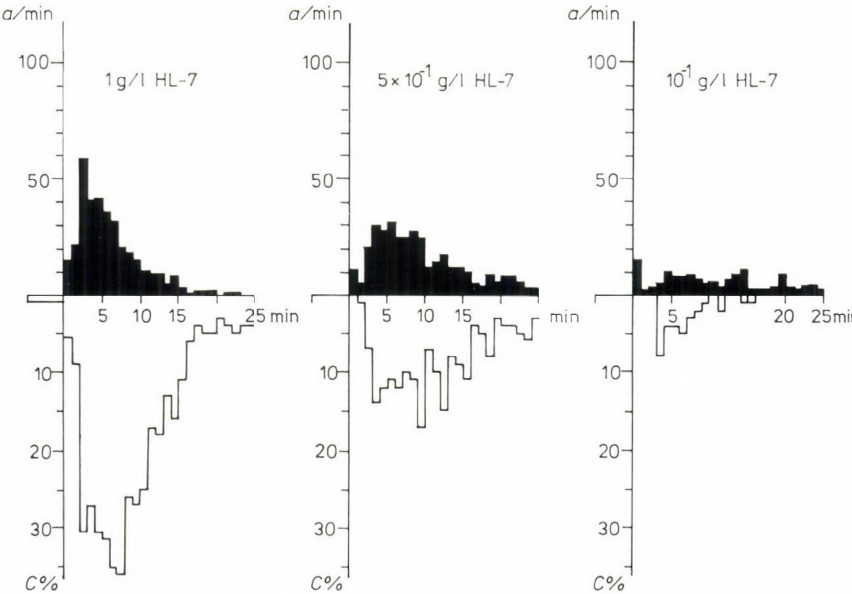

Fig. 1. Effect of Hungaria L-7 on the adductor muscle activity of glochidia. Ordinate above: total number of contractions of 100 glochidia per min (a/min) and below: percentage of larvae with lasting closure ($c\%$). Abscissa: exposure time in min

The TA-induced activity and its reduction by different concentrations of HL-7 are shown in Fig. 3*A* on mixed TA-sensitive population of glochidia. Figure 3*B* shows the percentage inhibition of TA-induced activity as a function of the concentration of HL-7 and its active ingredient lindane. On the basis of the curve, 50 per cent inhibition, of TA-induced activity is caused by 10^{-1} g per l HL-7, which is equivalent to 8×10^{-3} g per l lindane.

Dimecron-50 (D-50), if applied alone, did not show any significant effect in concentrations lower than 1 ml per l. In concentrations higher than 10^{-3} ml per l, it had an inhibitory effect on the TA-induced adductor muscle activity (Fig. 4). 50 per cent inhibition was observed at 6×10^{-1} ml per l concentration. This is equivalent to 3×10^{-1} ml per l active ingredient phosphamidon.

Since phosphamidon is known as a cholinesterase inhibitor, its effect was examined on acetylcholine (ACh)-induced activity, too; 50 per cent inhibition was caused by 2×10^{-1} ml per l D-50.

Thimet-10 G (Th-10) if applied alone had an effect on the adductor muscle activity of glochidia only in 5×10^{-1} g per l or higher concentrations. This slight excitatory effect was not more than 3–4 per cent of that of TA. Over 10^{-2} g per l concentration Th-10 inhibited the TA-induced activity (Fig. 5), and there was no significant difference between TA-sensitive and TA-insensitive populations. The 50 per cent inhibition of the TA-induced activity was observed at 8×10^{-2} g per l Th-10, which is equivalent to 8×10^{-3} g per l active ingredient phorate.

In concentrations lower than 1 g per l, Novenda applied alone did not show an excitatory effect on the adductor muscle activity of glochidia. In concentrations higher than 10^{-4} g per l it had an inhibitory effect on the TA-induced activity and 50 per cent inhibition was caused by a concentration of 2×10^{-1} g per l which is equivalent to 5×10^{-2} g per l active ingredient DNOC. Any difference between TA-sensitive and TA-insensitive populations could not be demonstrated (Fig. 6).

Bi-58 EC had some effect on the adductor muscle activity of glochidia only at concentrations higher than 5×10^{-1} ml per l. In the first few minutes following treatment, adductor muscle contractions occurred. The number of these contractions was only about 1–2 per cent of the TA-induced ones. At the same time, a great percentage of larvae (50 per cent at a concentration of 5×10^{-1} ml per l and 80–90 per cent at a concentration of 1 ml per l) showed lasting closure. In a 10^{-4} ml per l concentration, Bi-58 inhibited the TA-induced activity; 50 per cent inhibition was observed at 4×10^{-2} ml

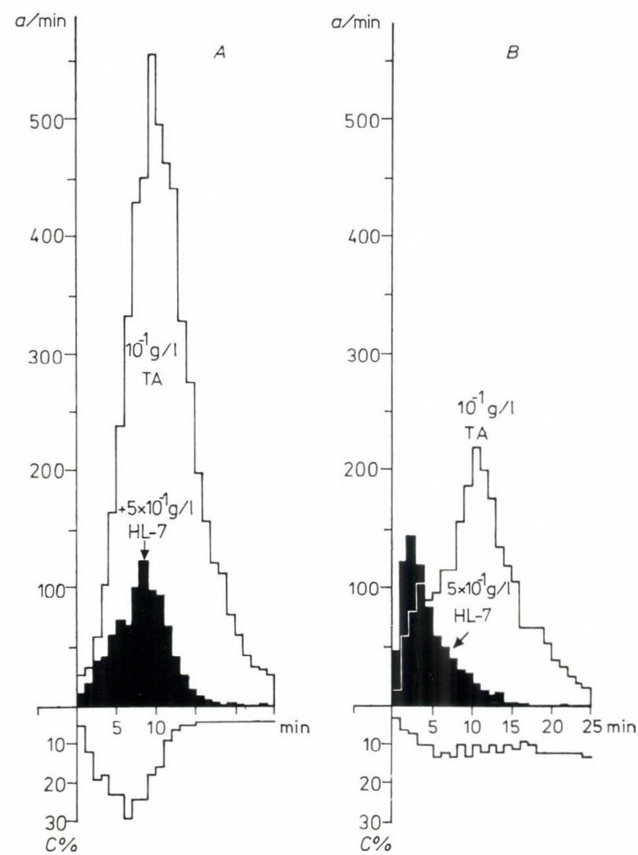

Fig. 2. Adductor muscle activity induced by 100 mg per l concentration TA and its inhibition by 5×10^{-1} g per l HL-7 on a TA-sensitive (*A*) and TA-insensitive (*B*) population. Ordinate below: percentage of the closed larvae at inhibition of the TA-induced activity by HL-7

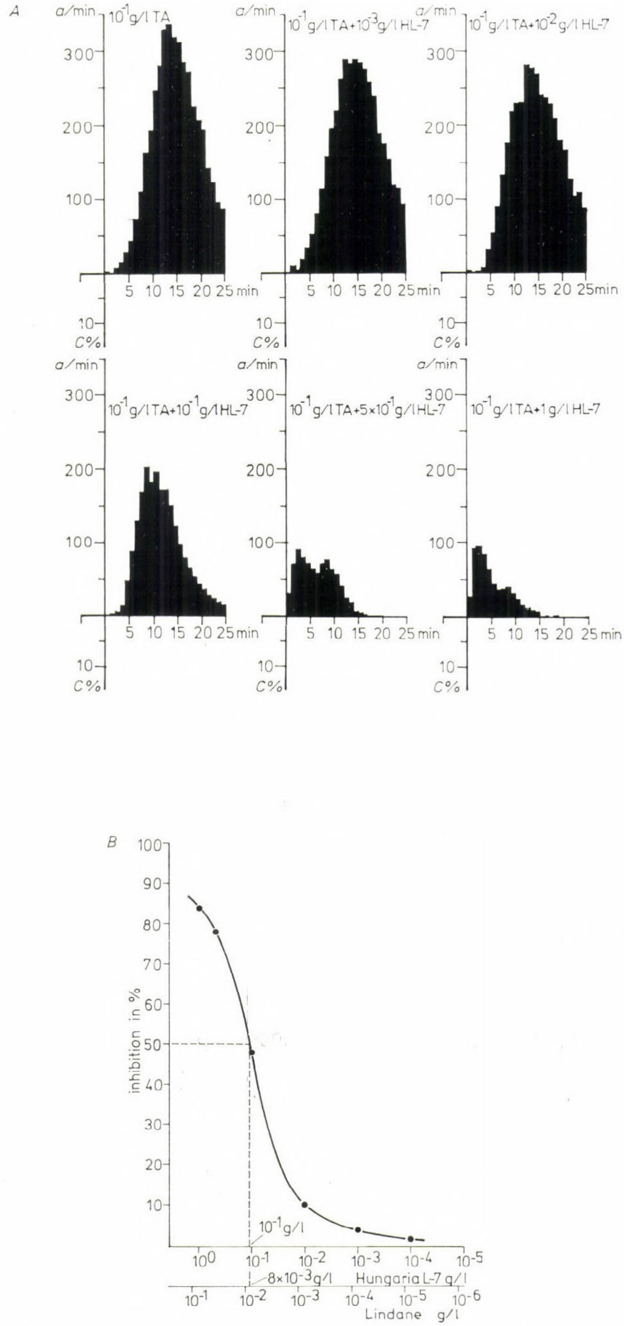

Fig. 3. Decrease of the TA-induced activity in the presence of different concentrations of HL-7 (*A*) and dependence of the inhibition of HL-7 concentration (*B*) in a population of glochidia of mixed sensitivity

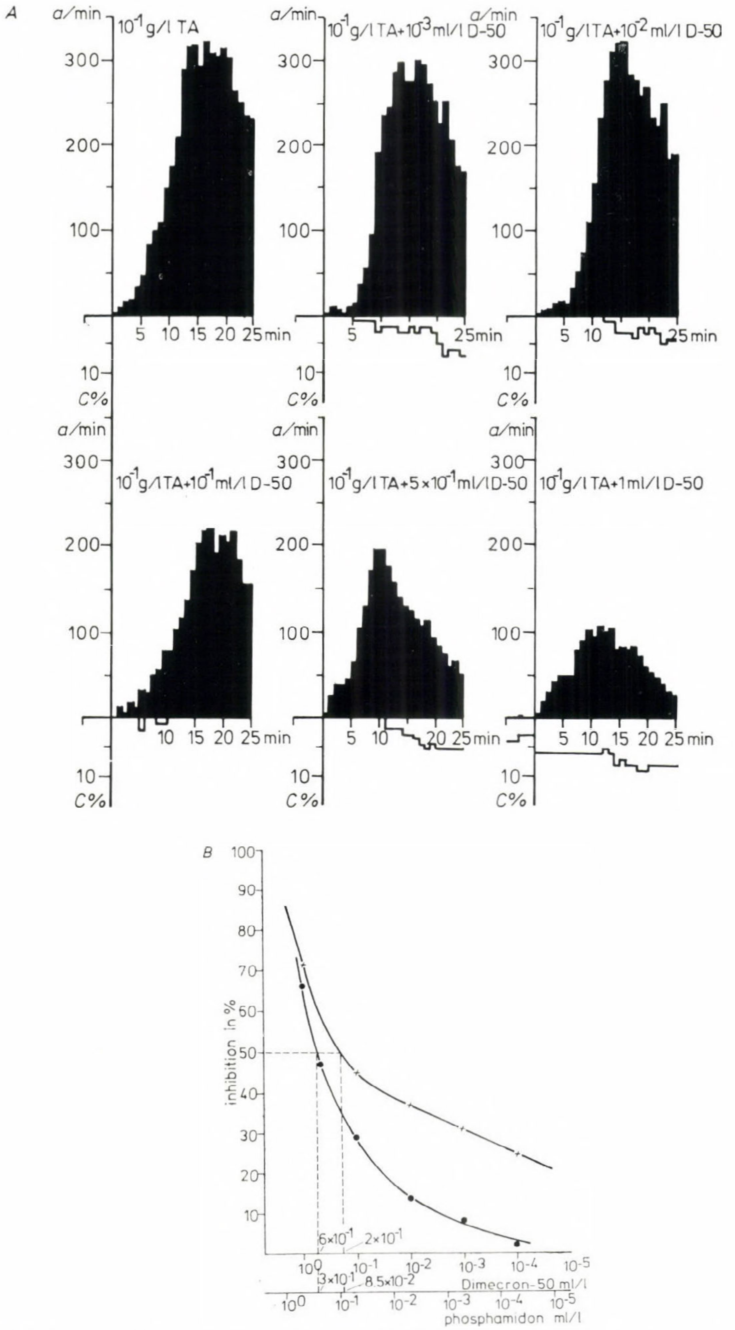

Fig. 4. Effect of different concentrations of Dimecron-50 on TA-induced activity (*A*) and dependence of inhibition on concentration (*B*) in a population of mixed sensitivity —·—·— TA-induced activity and —+—+—+ ACh-induced activity

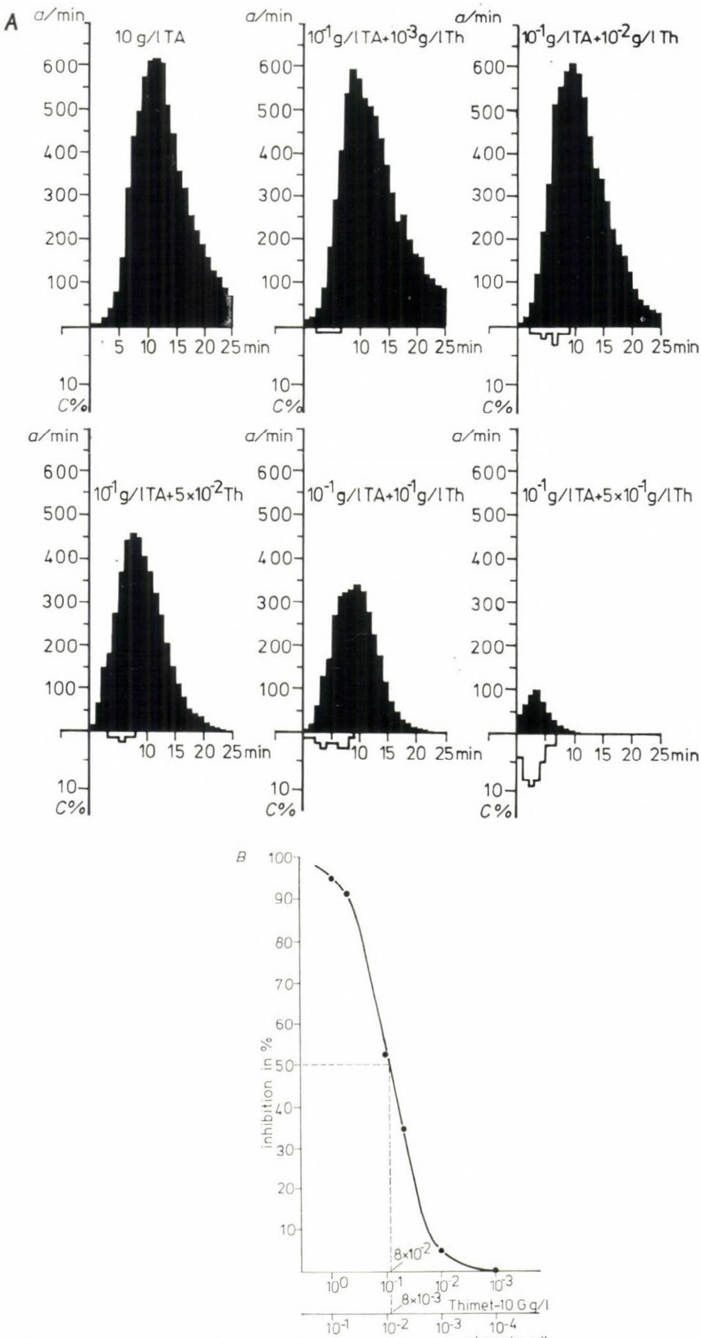

Fig. 5. Effect of Thimet 10-G on the TA-induced activity (*A*) and dependence of inhibition on concentration (*B*) in a population of mixed sensitivity

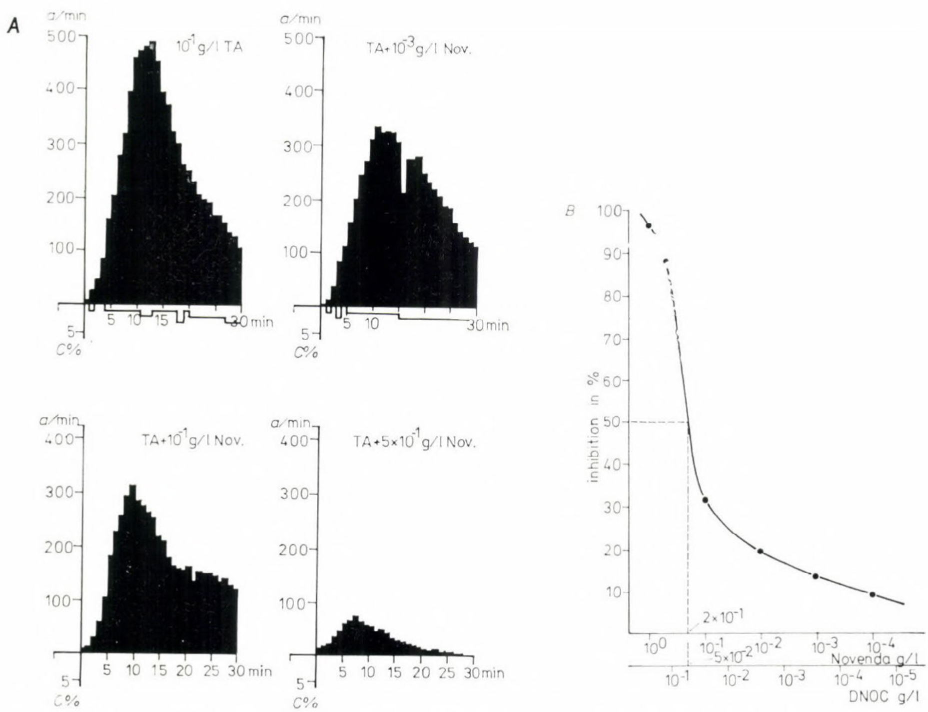

Fig. 6. Effect of Novenda on the TA-induced activity (*A*) and dependence of inhibition on concentration (*B*) in a population of mixed sensitivity

per l concentration, which is equivalent to 10^{-2} g per l active ingredient dimethoate. As in a concentration of 1 ml per l, Bi-58 prevents the excitatory effect of TA, the early concentrations and lasting closures of glochidia seem to be due to the direct effect of Bi-58 (Fig. 7).

In 10^{-1} ml per l or higher concentrations Malathion applied alone also had a slight excitatory effect on the rhythmic adductor muscle activity of glochidia, and 20–30 per cent of them exhibited lasting closure. The treatment with a concentration of 10^{-5} ml per l already resulted in a strong inhibition of the TA-induced activity, and 50 per cent inhibition occurred at 1.8×10^{-3} ml per l Malathion (Fig. 8). As the 10^{-1} ml per l concentration of Malathion prevented TA-induced activity, the contractions and lasting closure of glochidia were attributed to the direct effect of Malathion.

The effect of Malathion was also investigated after 24 hours' pretreatment of glochidia at a temperature of $+4$ to $+5$ °C; then, exchanging these solutions for TA-containing ones, the TA-induced activity was examined. The TA-induced activity of glochidia pretreated in this way was inhibited already by 10^{-8} ml per l Malathion, and 50 per cent inhibition was observed at 7.5×10^{-6} ml per l concentration (Fig. 9).

In concentrations higher than 10^{-1} ml per l the nematocide Shell-DD had a weak, short-term excitatory effect on the adductor muscle activity of the glochidia, while lasting closures occurred in 30–70 per cent depending on the concentration. Over 10^{-4} ml per l concentration it inhibited the TA-

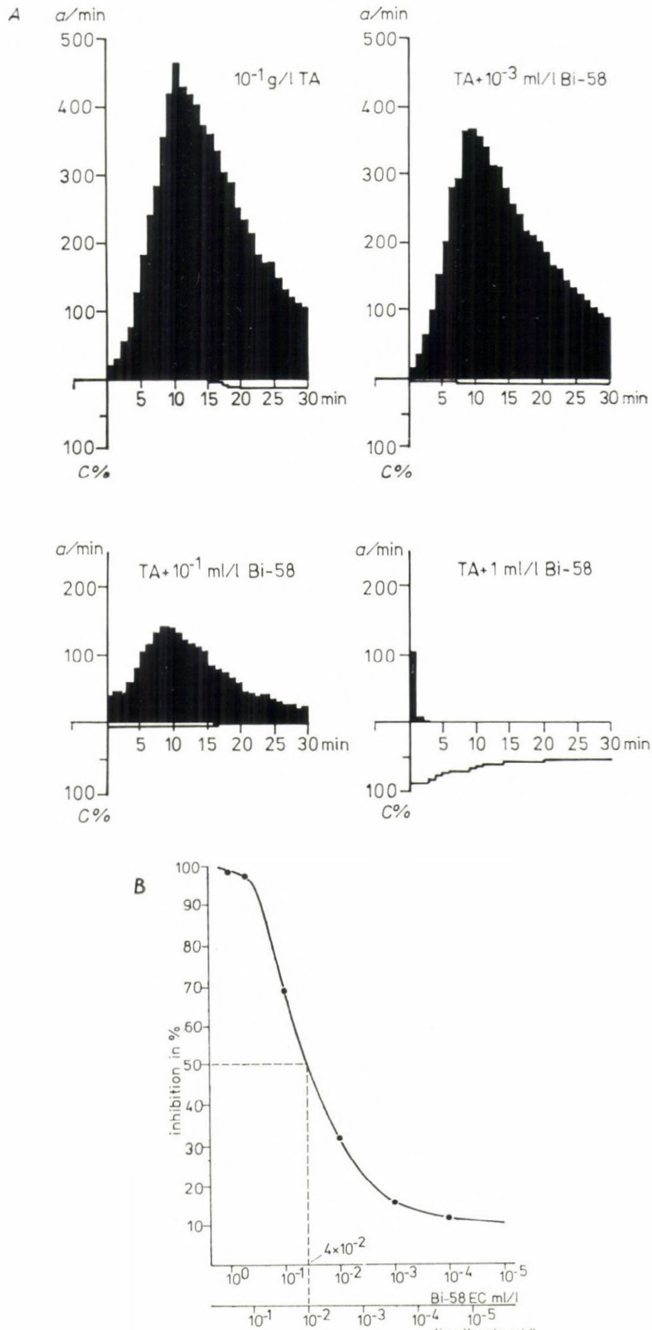

Fig. 7. Effect of Bi-58 EC on the TA-induced activity (*A*) and dependence of the in hibition on concentration (*B*) in a population of mixed sensitivity-

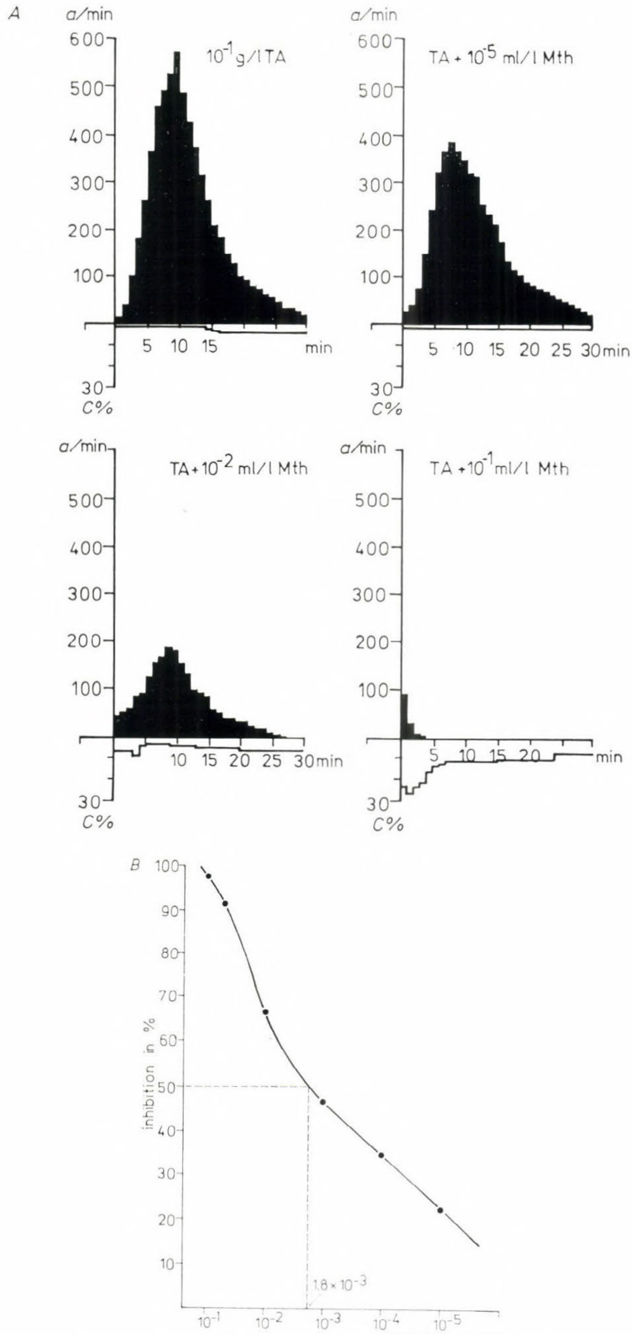

Fig. 8. Effect of Malathion of TA-induced activity (*A*) and dependence of the inhibition on concentration (*B*) in a population of mixed sensitivity

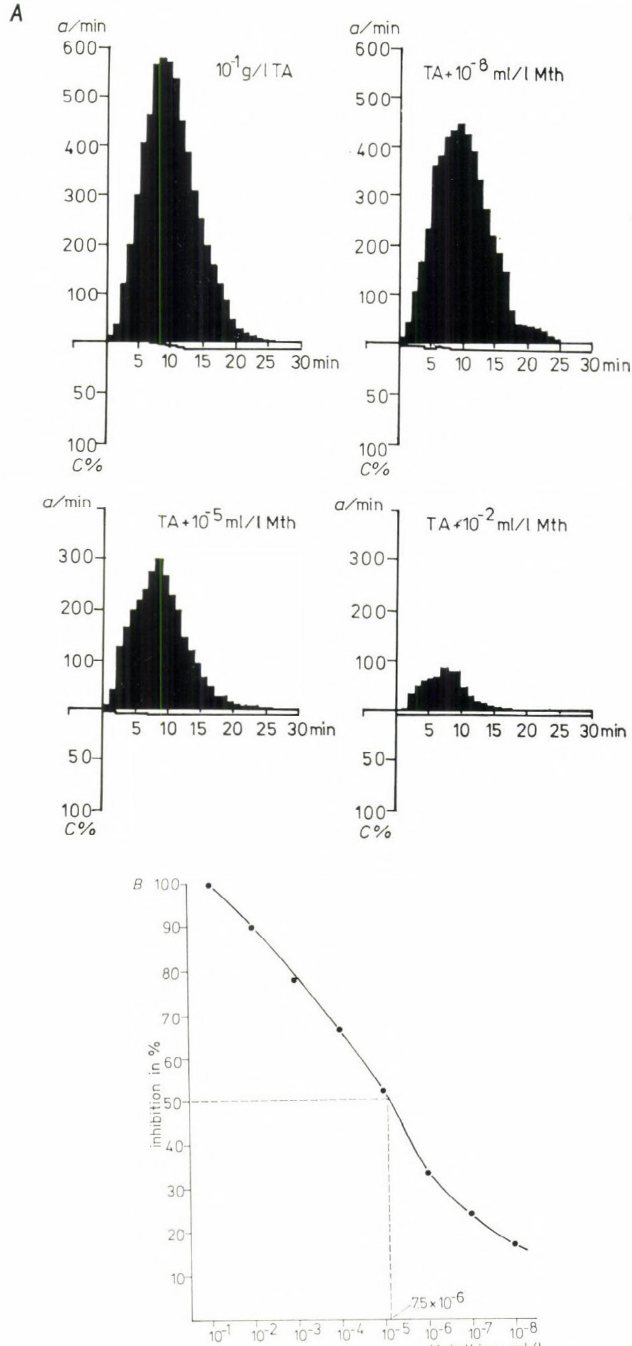

Fig. 9. Effect on the TA-induced activity of 24 h pretreatment with Malathion (*A*) and dependence of the inhibition on concentration (*B*) in a population of mixed sensitivity

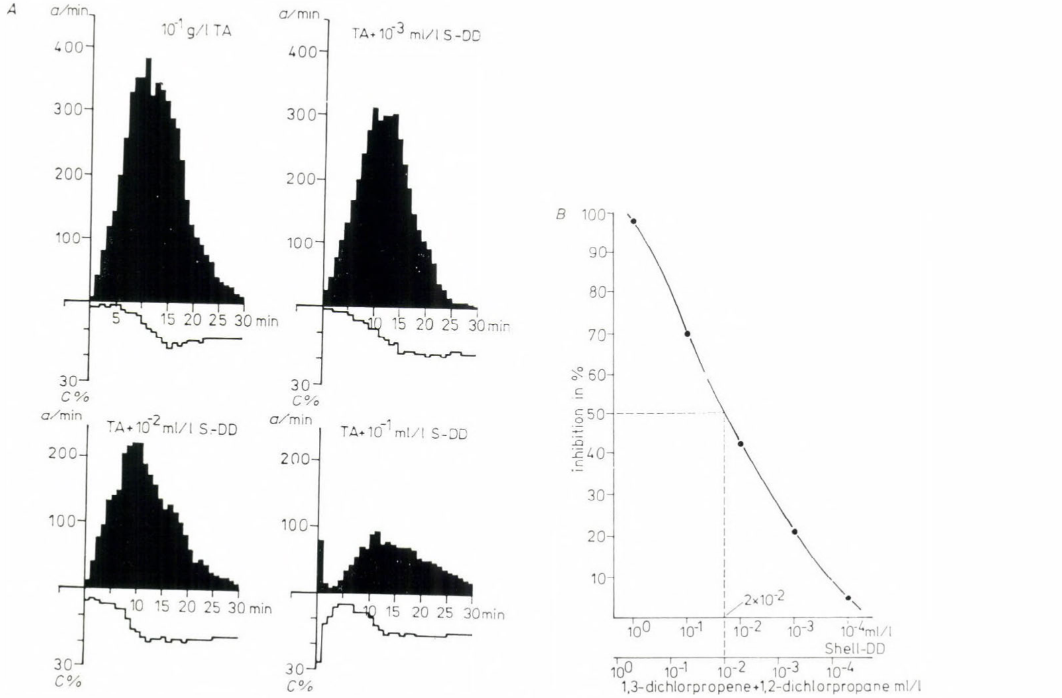

Fig. 10. Effect of Shell-DD on TA-induced activity (*A*) and dependence of the inhibition on concentration (*B*) in a population of mixed sensitivity

induced activity and prevented it in 1 ml per l concentration (Fig. 10). 50 per cent inhibition of the TA-induced activity was caused by 2×10^{-2} ml per l concentration, which is equivalent to 10^{-2} ml per l active ingredient.

The herbicide Gramoxon in a 1 ml per l concentration had a slight, short-term excitatory effect on the rhythmic adductor muscle activity of the glochidia, and 20–35 per cent of them showed lasting closure. In a concentration of 10^{-3} ml per l Gramoxon had a slight excitatory effect, while in concentrations higher than 10^{-1} ml per l it had an inhibitory effect. 10 ml per l concentration prevented the TA-induced activity (Fig. 11). 50 per cent inhibition was observed at 2.5×10^{-1} ml per l concentration, which is equivalent to 6.5×10^{-2} g per l active ingredient paraquat-dichloride.

Dikonirt in 5×10^{-1} g per l and higher concentrations had a slight, short-term excitatory effect on the rhythmic adductor muscle activity of glochidia and they often showed lasting closure (about 70–90 per cent). At a 10^{-1} g per l concentration Dikonirt inhibited the TA-induced activity (Fig. 12) and in a 5 g per l concentration completely prevented it. The effect of Dikonirt inducing lasting closure does not change in the presence of TA. The 50 per cent inhibition of TA-induced activity was observed at 3.5×10^{-1} g per l Dikonirt, which is equivalent to 2.5×10^{-1} g per l active ingredient 2,4-D.

The effect of pure 2,4-D (active ingredient of Dikonirt) was investigated, too. If applied alone, it was only effective in concentrations higher than

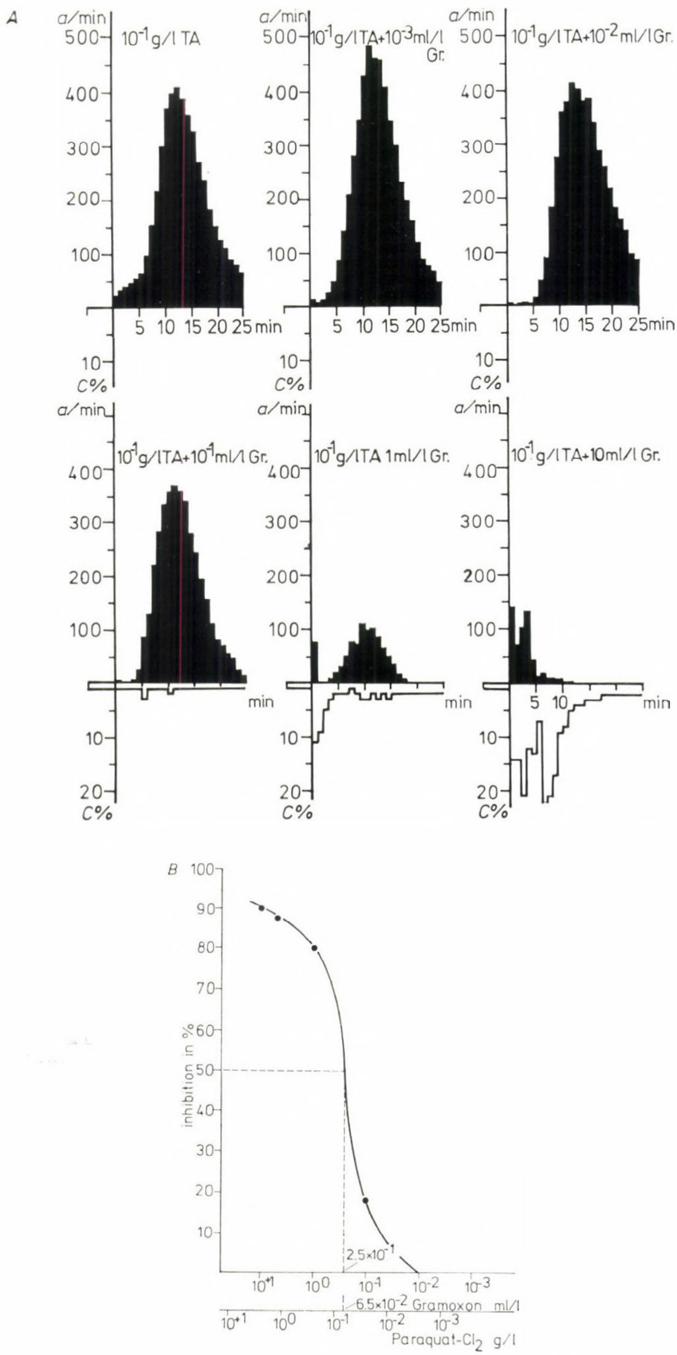

Fig. 11. Effect of Gramoxon on TA-induced activity (*A*) and dependence of the inhibition on concentration (*B*) in a population of mixed sensitivity

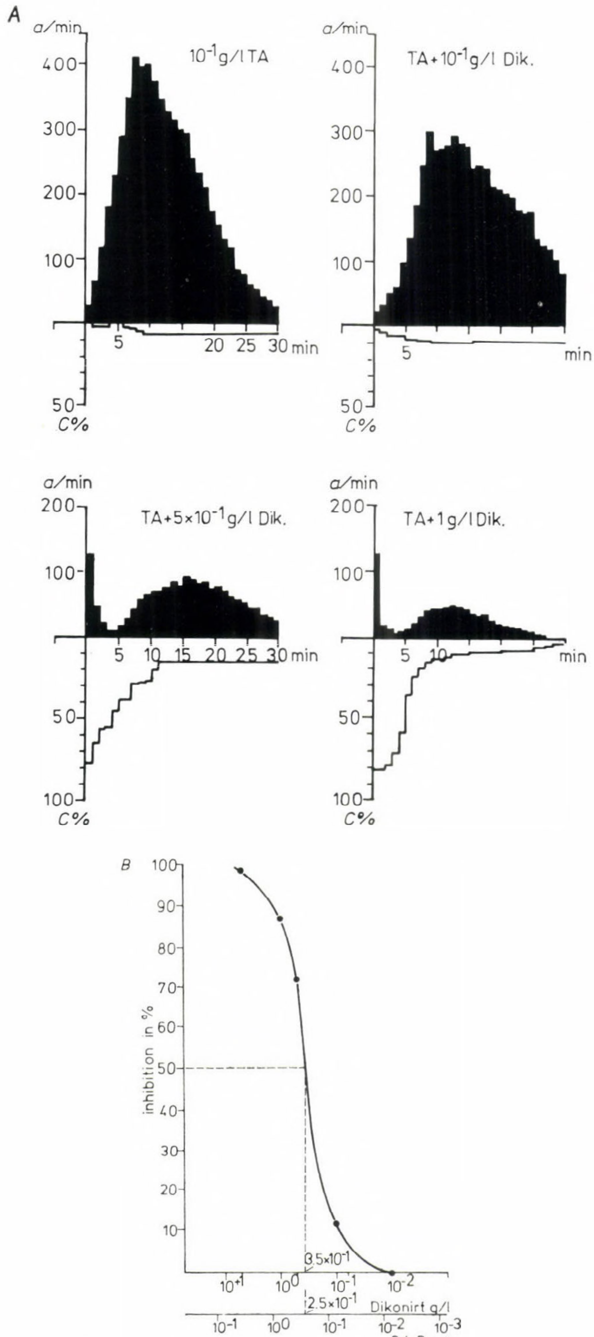

Fig. 12. Effect of Dikonirt on TA-induced activity (*A*) and dependence of the inhibition on concentration (*B*) in a population of mixed sensitivity

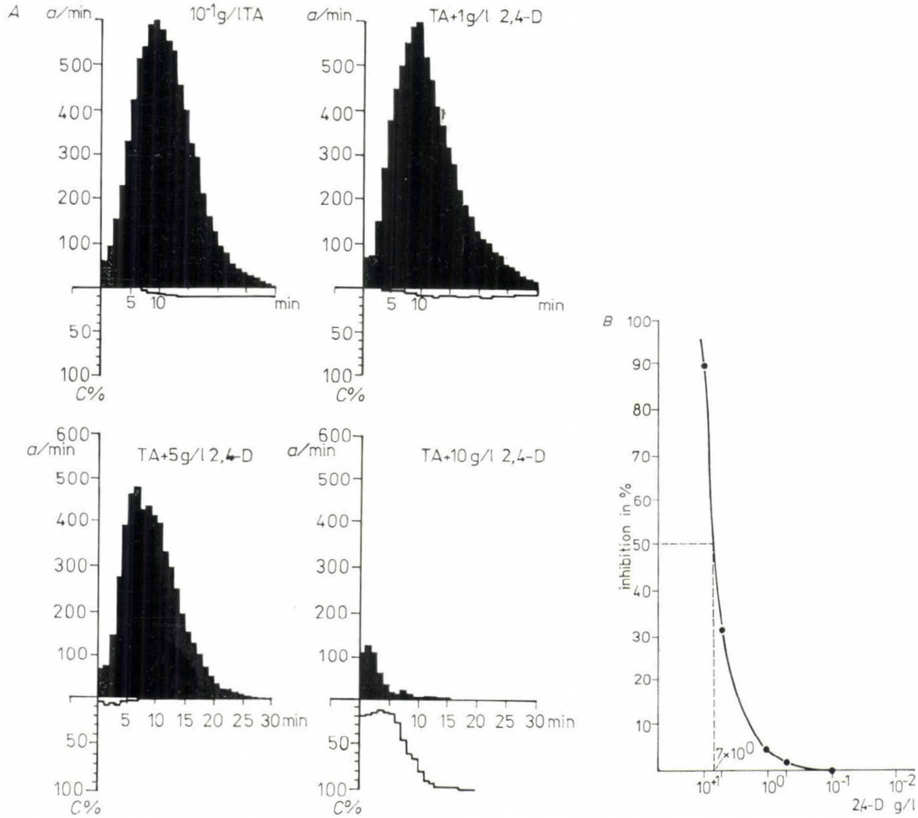

Fig. 13. Effect of pure 2,4-D on TA-induced activity (*A*) and dependence of the inhibition on concentration (*B*) in a population of mixed sensitivity

5 g per l. Its excitatory influence on the adductor muscle activity lasted 10–15 min, but after a five-min treatment the number of closed larvae increased, and reached 90 per cent in 15 min. 2,4-D had an inhibitory effect on the TA-induced adductor muscle activity only over a 1 g per l concentration (Fig. 13), and in 10 g per l concentration completely prevented it and only the direct effect of 2,4-D remained. The 50 per cent inhibition of the TA-induced activity was caused by a 7 g per l concentration.

If applied alone in a 10^{-1} g per l concentration, the surface active agent LAS had a slight short-term excitatory effect on the rhythmic adductor muscle activity of glochidia. At the same time, its effect inducing lasting closure was very strong, occurring in 80–90 per cent of the larvae within some minutes (Fig. 14). 10^{-1} g per l concentrations of LAS increased the total number of the TA-induced concentrations. The effect of LAS inducing lasting closure considerably decreased in the presence of TA. In 10^{-2} to 10^{-5} g per l concentration, LAS decreased the frequency of the TA-induced activity but increased its duration and the total contraction number.

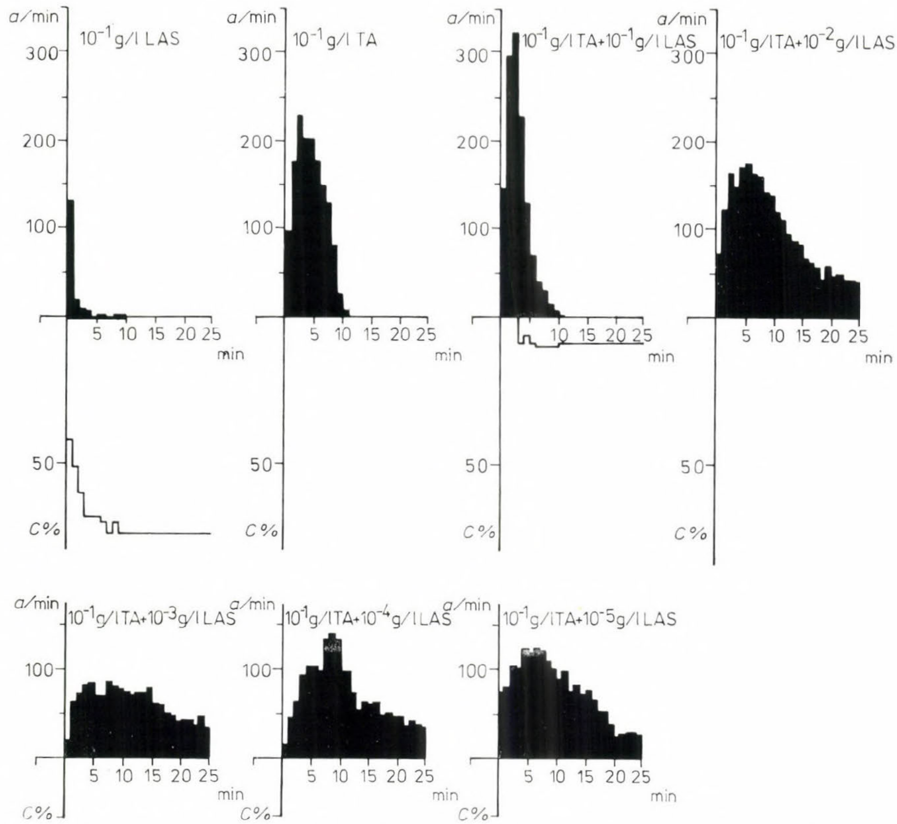

Fig. 14. Effect of LAS on the adductor muscle activity of glochidia and on their TA-induced activity in a TA-insensitive population

DISCUSSION

The effect of TA on glochidia is considered to be specific, and the presence of TA in the larvae has been suggested (S.-Rózsa and Lábos 1967). The presence of an adrenergic excitatory system is probable (Lábos 1970), and the occurrence of adrenergic neuromuscular synapses has been suggested (Zs.-Nagy and Lábos 1969). The weak excitatory effect of ACh on the adductor muscle activity of glochidia has been attributed to a non-specific effect (Lábos et al. 1964*c*, Lábos 1966).

As the spontaneous adductor muscle activity of glochidia generally has a low frequency (0–0.02 cpm), it is only suitable for testing excitatory effects. Only some of the tested compounds caused excitation in a high concentration: HL-7 ($> 10^{-1}$ g per l), Thimet-10 G ($\geqslant 5\times10^{-1}$ g per l), Bi-58 EC ($\geqslant 5\times10^{-1}$ ml per l), Malathion ($\geqslant 10^{-1}$ ml per l), Shell-DD ($\geqslant 10^{-1}$ ml per l), Gramoxon ($\geqslant 1$ ml per l), Dikonirt ($\geqslant 5\times10^{-1}$ g per l), 2,4-D salt ($\geqslant 5$ g per l) and LAS ($\geqslant 10^{-1}$ g per l). However, this excitatory effect was slight, and lasted only for a short time (several minutes) because lasting closure of a high percentage of the larvae occurred. It may be suggested that the excitatory effect of the examined pesticides on the adductor muscle activity of glochidia is caused by a non-specific, but strong, toxic influence which is lethal in most of the cases. The lasting closure of the larvae may be a result of this effect.

Different concentrations of the examined pesticides had an inhibitory effect on the TA-induced adductor muscle activity of glochidia. The concentration reducing the total number of TA-induced contractions by 50 per cent seems to be a good index of the inhibitory effect of toxic compounds (this is called 50 per cent inhibition). These concentrations and the toxicity sequence are shown in Table 1.

The pesticide concentrations causing 50 per cent inhibition ranged between 1.8 and 600 ppm. These concentrations are so high that their occurrence is not probable in natural waters. Nevertheless, it must be considered that the larvae were exposed to the toxic effect of the examined concentrations of pesticides for 30 min altogether. An increase in the exposure time leads to a decrease in toxic concentrations as shown in the case of Malathion: an increase in exposure time from 30 min to 24 h resulted in the decrease of the concentration required for 50 per cent inhibition from 1.8 ppm to 0.0075 ppm. Under natural circumstances glochidia may survive for a long time in polluted water containing these pesticides in concentrations lower than had been used in our experiments.

No comparative data are available in the literature concerning the toxicity of pesticides on the larvae of fresh-water mussels. Approximately 60 per cent of the eggs of marine clam (*Venus mercenaria*) and 43 per cent of eggs of *Crassostrea virginica* developed normally in concentrations of lindane up to 10^{-2} g per l (Davis 1961). The 48 h TL_m-value (concentration at which 50 per cent of the eggs developed normally or 50 per cent of the larvae survived) was found to be over 10 ppm for clam eggs and 9.1 ppm for oyster eggs, however, the 12-day TL_m-value on clam larvae exceeded 10 ppm (Davis and Hidu 1969). All these values are near the concentration of the active ingredient of HL-7 (8 ppm lindane) which, in our experiments, decreased the total number of TA-induced concentrations within 30 min exposure time.

The 48 h TL_m-value of Malathion on oyster eggs was described to be 9.07 ppm, while the 14-day TL_m-value on larvae was 2.66 ppm (Davis and Hidu 1969). In the present experiments on *Anodonta* glochidia the 50 per cent inhibition of TA-induced activity was caused by 1.8 ppm Malathion at 30 min exposure time and by 7.5 ppb Malathion at 24 h exposure time.

1 ppm concentration of the 2,4-D salt stimulated the growth of the oyster larvae, however the 48 h TL_m-value for the eggs was found to be 20.44 ppm and the 14-day TL_m-value for the larvae was 64.29 ppm (Davis 1960; Davis and Hidu 1969). These values are much lower than the concentration of 7000 ppm which resulted in 50 per cent inhibition of the TA-induced activity of glochidia in the present experiments.

0.05 ppm LAS already inhibits the fertilization of *Mytilus edulis* eggs and the 240 h growth of the larvae is reduced by concentrations higher than 0.1 ppm (Granmo 1972). Although 0.01 to 10 ppm concentration of LAS inhibited the frequency of the TA-induced activity of glochidia, it increased the duration of this activity and the total contraction number.

TABLE 1

Pesticide Concentrations Causing 50 per cent Inhibition of the TA-induced Activity of Glochidia

Compounds	Active ingredient	Inhibitory concentration (ppm)
Insecticides		
Hungaria L-7	7.8 per cent lindane (γ-HCH)	100
Dimecron-50	50 per cent phosphamidon	600
Thimet-10 G	10 per cent phorate	80
Novenda	25 per cent DNOC (4,6-dinitro-*o*-cresol)	200
Bi-58 EC	25 per cent dimethoate	40
Malathion	98 per cent pure	1.8 0.0075*
Nematocide		
Shell-DD	50 per cent 1,3-dichlorpropene* 1,2-dichlorpropane	20
Herbicides		
Gramoxon	40 per cent paraquat-dichloride	250
Dikonirt	75 per cent 2,4-D (2,4-dichloro-phenoxyacetic acid)	350
2,4-D	96.5 per cent pure, 2,4-D salt	7000

* At 24 h exposure time.
Effectiveness: Malathion > Shell-DD > Bi-58 EC > Thimet-10 G > Hungaria L-7
Novenda > Gramoxon > Dikonirt > Dimecron-50 > 2,4-D salt

It is well known that pesticides often contain activators and synergists in addition to the active ingredient. In these cases, the toxicological test of the pure active ingredients will not suffice, and the testing of the compounds actually used is necessary. In our experiments the herbicide Dikonirt proved to be more toxic than pure 2,4-D salt which is its active ingredient.

The effect of pesticides may be lethal either due to their direct toxicity or as a result of their interference with functional conditions which are prerequisites of the ontogenic development of glochidia. Thus the inhibition of the rhythmic adductor muscle activity or the induction of lasting closure stops the ontogenic development of larvae because for their sticking to the epidermis of fish (a condition of their metamorphosis) rhythmic adductor muscle activity is needed. In the absence of adherence, ontogenic development is interrupted and the larvae will perish. Decrease in the Pelecypoda fauna of streams and stagnant waters can be attributed to this effect in some cases. The reduction affects the natural purification process of freshwaters adversely due to a lower filtration activity.

SUMMARY

Studies on the toxicity of some pesticides on the larvae (glochidia) of the fresh-water mussel *Anodonta cygnea* L. have shown that the spontaneous rhythmic activity of larvae cannot be used as an index of the effect of inhibitory compounds because of the low frequency of their spontaneous activity. However, the TA-induced adductor muscle activity of glochidia might be suitable for this purpose.

The insecticides, nematocide and herbicides tested on the glochidia has been found to differ as regards toxicity. The inhibitory concentration resulting in 50 per cent inhibition of the number of TA-induced contractions ranged between 1.8 and 600 ppm.

Increasing the 30 min exposure time the effective inhibitory concentrations decreased, indicating that the sensitivity of this method may be higher.

The toxicological test of the pure active ingredient is not enough, and the compounds actually used must be invariably tested because the different synergistic and activator effects will interfere with the end results.

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DEEP WATER DEPLETION AS A METHOD FOR OLIGOTROPHICATION OF LAKES

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Abstract

Hypolimnetic depletion means the decreasing of nutrients in a lake, i.e. a process of artificial oligotrophication. Accordingly it provides a useful way for restoration of stratified lakes.

Since 1976, Lake Arendsee (G.D.R., maximum depth 49.5 m) has only had an artificial outflow through which all excess water is taken out from the deepest water layer. The increased export of nutrients by hypolimnetic depletion is expected to be fivefold for phosphate and 23-fold for inorganic nitrogen.

Diminished productivity creates better living conditions for salmonids (higher O_2 content in the deep water layers) and is also more favourable from the aspect of recreation (clean water with higher transparency).

INTRODUCTION

A stratified lake with natural outflow becomes a trap for many substances important in the turnover of nutrients. This happens particularly to plant nutrients like phosphorus and nitrogen, accumulated in high concentrations in the hypolimnetic layers during the stagnation periods. The natural outflow affects only the epilimnic layers being the poorest in nutrients. During the circulation periods the lake is fertilized internally and the annual cycle of productivity is continued at the same or at an even higher level. The aim of deep water depletion is to reverse this process into the opposite direction. In this case an artificial oligotrophication can be achieved inducing better oxygen conditions in the deep water layers.

Lake Arendsee (Magdeburg district, G.D.R.) is going to be restored following the example of Lake Kortowo in Poland (Olszewski 1951, 1973).

RESULTS AND DISCUSSION

The surface area of Lake Arendsee is 5.5 km^2 with a mean depth of 30 m, the maximum depth being 49.5 m. The lake is a favoured resort area. From the point of view of fisheries, salmonids (particularly *Coregonus albula*) are of the greatest importance. Because the eggs of this species develop at the bottom, the preservation of aerobic conditions in the bottom-near water layers is badly wanted.

Since 1970 the lake has been freed from all kinds of sewage load. The drainage systems of the town Arendsee and of its dairy industry are treated at a sewage plant. In summer, the purified sewage is used for irrigation of pastures, and in winter its load is diverted into another catchment area. Nevertheless, a fast improvement in the quality of water is not to be expected, because the exchange rate of the lake water is only 3.6 per cent yearly. Dilution in a fully mixed reactor is going on according to the following equation:

$$C = C_0 \cdot e^{-Dt}$$

where C_0 is the initial concentration of a substance at the time t_0, D is the rate of renewal, t the dilution time, and C is the concentration which may be expected after t years, provided that no further substance reaches the lake. The 'half-life' of dilution in Lake Arendsee is 19 years. At the end of the summer stagnation the phosphate content at the deepest point of the lake is eight times higher than in the surface layers, while the mineral nitrogen content is about 23 times higher.

This means that the nutrient output can be considerably increased by means of hypolimnetic water depletion. The realization of this project has been hitherto opposed to because of the expensive technology of laying the conventional submerged piping system. Recently, however, new spiral pipes of low specific gravity have become available which may be attached to floating bodies in the lake. In the required position, just above the deepest point of the lake, the holding strings are severed, the pipe is slowly lowered to the bottom of the lake and, being flexible, is accommodated to the profile of the bottom. In the same manner, the connection on the shore may be attached to the underground steel or concrete conduits or rather to a reservoir used as a receiver for the irrigation plant.

In the early summer of 1976, the outflow of Lake Arendsee from 49 m depth was built. Three polyethylene pipes, each 300 m long, with 500 mm nominal diameter, were laid from the natural outlet offshore down to the deepest point. The original surface outflow was closed so that hypolimnetic water depletion could be regulated by slide valves. At this point a venturi flume and an equipment for measuring the quality of water were installed to get exact information on the dilution process in the lake. Since the apex of the pipe is below the water surface (approx. 0.5 m), the outflow takes place automatically without additional expense of energy. Owing to a storage-type outflow regulation it is possible to satisfy the demand for irrigation water in the summer season. Considering the greatest possible efficiency of nutrient outflow, the delivery is favourable in summer. The increase of nutrient output by hypolimnetic depletion is expected for phosphate to be approximately fivefold and for inorganic nitrogen to be 23-fold.

Along with oligotrophication, the recreation value of Lake Arendsee will be improved. The expenses for the investment will be compensated after a few years by the profit obtained on the recreation side.

The above example of hypolimnetic water depletion at Lake Arendsee has had a stimulating effect on further restoration projects in the G.D.R.

SUMMARY

On Lake Arendsee (G.D.R., maximum depth 49.5 m) restoration has been going on by deep water depletion since 1976. The aim of this process has been to decrease the nutrient concentration and bioproductivity and thus to stimulate the oligotrophication of the lake. Lake Arendsee is an important recreational centre and has some fish yield (e.g. salmonids). The entire outflowing water mass is taken out from the deep layers of the lake through flexible plastic pipes. The increase of nutrient export by hypolimnetic depletion is expected to be fivefold for PO_4 and 23-fold for inorganic N.

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THE APPEARANCE OF A LITTORAL FAUNA AND FLORA IN LAKE NASSER–NUBIA, A NEWLY FORMED TROPICAL MAN-MADE LAKE

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Abstract

The process of the appearance of a unique succession of terrestrial and aquatic forms under desert conditions is described. On the bare rocks and empty sandy beaches a rich, newly formed plant association has appeared composed of a few species, being, however, rich in specimens. Besides, a water-bound fauna has developed dominated by chironomids, water bugs, spiders, toads, underwater epilithic algae and fish-fry (mainly *Tilapias*). These have been followed by rooted aquatic weeds, molluscs, dragonflies, shrimps and oligochaetes and by typical terrestrial forms as locusts, butterflies, scorpions and birds, etc., on the shore. A basic influence of the seasons and of low- and high-water conditions could be demonstrated.

The River Nile, called 'the father of the rivers' in ancient times, carries the immense amount of roughly 75 km^3 water every year from the Ethiopian and Central African Highlands through the desert to Egypt since times immemorial.

The yearly flood of the river carries a heavy load of sand and silt, about 100 million tons, which has settled all along the Egyptian section of the Nile Valley, causing its proverbial fertility for thousands of years (Hurst 1957).

Cultivated land, abundant riverside vegetation and, in early times, swampy areas rich in waterfowl and wildlife surrounded the river where nowadays Lake Nasser and its Sudanese part, Lake Nubia, are situated. Following the construction of the Aswan High Dam the landscape underwent profound changes. All arable land with about 500 000 date palms, and all the accompanying plant and animal life had been flooded from Aswan southwards over a distance of about 500 km upstreams (Fig. 1). The huge reservoir thus produced was surrounded by bare rocky, stony and sandy shores of the mighty lifeless desert (Fig. 2).

During the activities of the UNDP Lake Nasser Development Centre Project between 1969 and 1974, studies were undertaken to follow up the reappearance and spreading of aquatic and terrestrial life in the littoral zone at rising and falling water levels as influenced by natural floods, and particularly by regulated waterflow through the spillways and the turbines of the High Dam, causing a strong combined effect on the water level fluctuations up to 5 or even 10 m annually (Entz 1974, Entz and Latif 1974). This caused the inundation of hundreds of square kilometres of desert area during the high-water period and their drying out during the low-water period.

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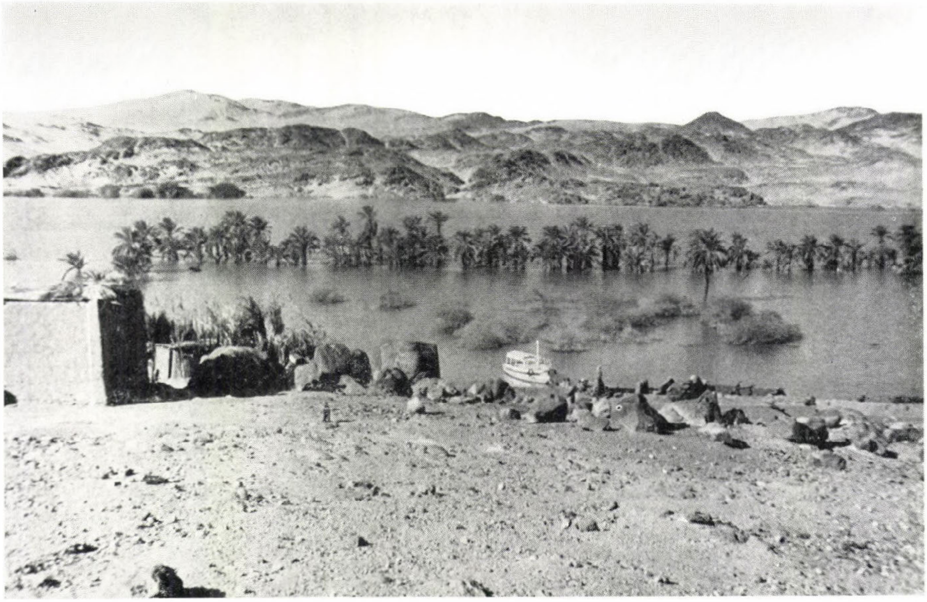

Fig. 1. Some of the half million date palms of Nubia, flooded by the lake in December 1972 (All photos are original, taken by the Author)

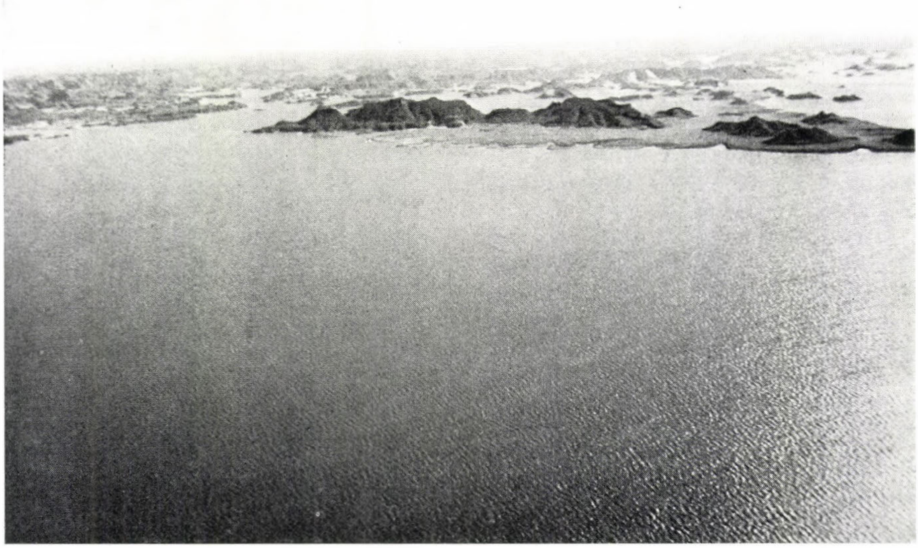

Fig. 2. Bare rocks and sandy beaches in the area of Ibrim (Lake Nasser, January 1974)

This unusual situation has been made still more peculiar by some unique conditions prevailing in the surrounding desert. The River Nile is the only source of water of the lake. Tributarie or drainage water from the adjacent area are absent. Rainfall is near nil; although the mean yearly precipitation is about 4 mm, several years may pass without a single drop of rain. Light conditions are almost uniform throughout the year because of the lack of clouds in this land of eternal sunshine, where even all kinds of human pollution are absent.

Yearly fluctuations of limnic conditions are affected by three main factors:

1. The flood of the Nile reaching the lake from the south in July, with highly turbid water showing low conductivity values. During the period of our investigations these water masses left the reservoir to the north within 6 to 12 months, after sedimentation of the suspended silt load.
2. The annual cycle of atmospheric temperature with mean values of 15 °C in January and 38 °C in July causing a yearly fluctuation of the mean water temperature ranging between 17 °C and 29 °C.
3. The wind blowing all year round, mainly from the north or northwest, and frequently causing a surface current in the lake from north to south, i.e. just opposite to the main water current.

The rising water level when first intruding into the desert washed a remarkable amount of soluble materials out of the obviously rich desert soil of the newly inundated region. Some of the dissolved salts ascended through the soil due to capillarity even above the highest water level, and formed there a solid crust after the aquatic solvent had rapidly escaped with an extremely high evaporation rate (8 mm per day in winter and 15 mm per day in summer).

Usually a crystalline white crust of almost pure NaCl formed; on this a brownish, highly hygroscopic layer of other salts was superimposed. Water, together with these considerable amounts of salt probably rich in nutrients, have possibly been the major factors promoting the early appearance and fast development of rich aquatic and terrestrial vegetation in the newly established littoral zones (Fig. 3).

Aquatic life appeared first in the form of epitectic, unicellular diatoms, green algae and blue-greens. These pioneer settlers were followed immediately by filamentous algae and shortly afterwards by well-known invaders of newly formed water bodies as Chironomidae and other insects grazing together with the cichlid fish *Tilapia* on the green mats of algae. Chironomid larvae were found in hundreds or even thousands per m^2 at 1 to 7 m depths of the littoral zone. Huge swarms of emerging adult midges appeared in an area already during the second or third year of its inundation. Similar phenomena could be observed in many other newly formed reservoirs like the extensively studied Volga cascade system (Mordukhai-Boltovskoi, in press). Later on, as usual, a decrease of the midges could be observed.

The inundated area grew gradually, and the centre of chironomid invasions moved southwards each year.

The development of chironomids was influenced by the floods in a peculiar way (Entz 1974, Rzóska 1976). Swarms of adults appeared in large numbers in the flood season at rising water levels, but always before the

Fig. 3. Salt crust caused by capillarity. Lower white crust, pure NaCl. Upper dark cover, mixture of hygroscopic salts (June 1973)

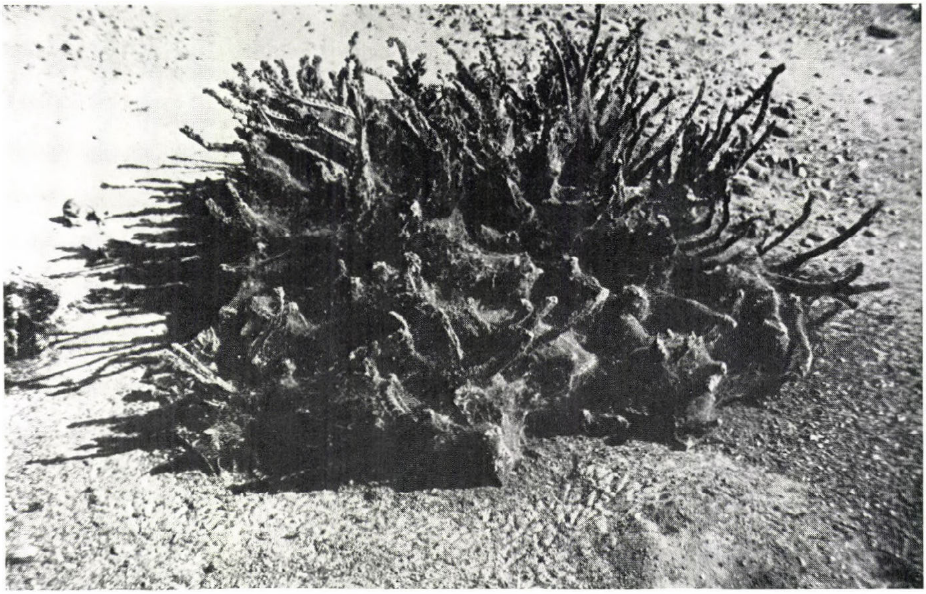

Fig. 4. Hyoscyamus muticus on the lake-shore covered by spider-webs heavily loaded with dead chironomid midges (February 1973)

arrival of the turbid flood water: e.g. in July 1970 at Abu Simbel (300 km south of the High Dam), with an increasing delay towards the northern sections of the lake and finally in December or January near the High Dam (Entz, in press).

Undoubtedly, this phenomenon was independent of the temperature and light conditions, both being generally uniform all over the lake at the times of observations during the season in question. The appearance of adults was possibly related to the peak in salinity reached in different areas just before the arrival of the turbid flood water. This is accompanied by increased hydrostatic pressure, which may be double or manyfold in the littoral zone during the upcoming flood. A number of questions concerning this peculiar habit of chironomids are still unanswered and should be studied in the future in detail.

Mixed with chironomids were larvae of a *Micronecta* sp. (Hemiptera) appearing in great numbers, similarly feeding on epitecton. A characteristic, almost uninterrupted rhythmic underwater chirping of the bugs in the night could be heard all along the shore.

These insects exerted a basic influence on the development of terrestrial animal life on the shore. The giant swarms of chironomids reached the shores after having deposited their eggs and settled mostly on wind-protected surfaces (rocks, sandbanks, plants, shelters, etc.), and died shortly thereafter, forming a cover of a foul-smelling thanatocenosis several mm, or even several cm thick (Fig. 4).

Spiders carried by flying gossamers to distances of several hundred km or spreading by other means settled on the shore and fed on the swarming chironomids. The number of spiders increased tremendously in the littoral zone within a few years. Both insects and spiders served as plentiful food for toads (*Bufo* sp.), which became similarly widespread within 3 to 6 years after the formation of Lake Nasser. At that time several adult toads could be counted per one meter shoreline in different areas.

The exuviae of midges and waterbugs together with detritus particles were drifted ashore and formed mats of decaying organic matter attracting flies and other insects, and were used as food as well as sites for propagation by these animals. In such sandy littoral areas the density of insects occasionally reached several hundred per m^2. About 7 years after the formation of the reservoir dragonflies became numerous (Fig. 5). Both the larvae and adults being carnivorous, they decimated together with fishes, spiders and toads, the stock of chironomids. As a consequence, the number of the latter decreased abruptly, followed by a similar decrease of their consumers, spiders flies and toads. This process was accompanied by profound changes in the aquatic and terrestrial life, mainly due to the appearance of macrovegetation both underwater and on the land.

During the first years of its existence, aquatic macrovegetation was practically absent from the reservoir. *Potamogeton pectinatus* appeared first and became widespread in 1972 from Aswan up to the Sudanese border but was still absent in Lake Nubia in 1974. *Potamogeton pectinatus* stands were spread only in shallow (0 to 2 m deep) water areas at decreasing water level (January–May), while they dried and died in the summer months.

At 3 to 7 m depths two other species (*Najas armata* and *N. minor*) appeared in the northern sections of the reservoir in 1973–1974, sometimes in

Fig. 5. Dragonflies at noon around a shallow rock-pool on the lake-shore at Wadi Halfa. All insects keep themselves in a vertical position possibly to reduce evaporation (August 1973)

dense populations, particularly near the High Dam. Floating buds or branches of all the mentioned aquatic weeds could be observed in the summer in several open-water areas and bays (called Khors) of Lake Nasser. It is worth mentioning that *Potamogeton crispus*, the most common species of Aswan Lake (a small reservoir between the Old Aswan Dam and the High Dam), was not found in Lake Nasser throughout 1974.

The macrovegetation became rich in epiphytic organisms and many littoral forms, like the freshwater shrimp *Caridina nilotica* (?) and the fry of different fish species living sheltered in the thickets of macrophytes. Shrimps were earlier unknown in this section of the Nile and have been first detected in the stomach contents of nileperch (*Lates niloticus*), the most famous fish of the Nile, in 1972 (Beadle 1974). During 1973–1974, shrimps became very abundant particularly in the macrophytic complexes of the lake. In 1973, another freshwater crustacean, a crab (*Potamonautes niloticus*) appeared in Lake Nasser. In some spots (e.g. in the area of Garf Hussein, 70 km from the High Dam), it appeared in great numbers but up to 1974 its distribution remained scattered.

Floating weeds, like the ill-famed *Eichhornia crassipes* (water hyacinth) and *Pistia stratoides* remained absent from the reservoir.

As aquatic life was affected remarkably by the appearance of aquatic weeds, the same happened to the terrestrial vegetation. The supralittoral zone, the shores of the lake, had been completely bare in the first years (Fig. 6). In 1970, millions of seedlings of a common weed, *Glinus lotoides* (Aizoaceae), grew on the shores during decreasing water level. The seeds of this extremely prolific weed were spread most probably by flood waters from south to north. The seedlings of the plants seem to be arranged into

Fig. 6. Freshly deposited silt near Wadi Halfa (Lake Nubia, September 1973)

obvious rows, parallel to each other at the water's edge (Fig. 7). This arrangement might be explained by a stepwise regression of the water level. This plant covering wide areas along the shore but being equally common on rocky and on sandy areas, became the most common macrophyte around the lake in 1970–1973. Spreading from the north *Rumex dentatus*, *Tamarix nilotica* and *Zygophyllum coccineum* became numerous. In flooded, but afterwards dried, localities sometimes bushy, park-like areas came into being on spots where vegetation had been completely absent for thousands of years (Fig. 8). Some plants appeared in spots, but all over the littoral area, like *Hyoscyamus muticus* (Solanaceae), a poisonous herb being the most typical erect herb in 1971–1973. Certain species were only found along the line of maximum water level, e.g. some desert elements like handal (*Colocynthis vulgaris*, Cucurbitaceae) having a pleasant scent but extremely bitter tasting fruits, or a spiny flowering plant, *Fagonia bruguieri* (Zygophyllaceae), or the white-flowered *Morettia philaene* (Cruciferae). The latter plants are typical examples of desert forms, appearing scattered where some humidity is present but unable to withstand inundation. Others — mainly riverside elements — are able to survive flooding for several months or probably even for more than a year (e.g. *Tamarix nilotica*).

During the last year of our research work, an increased spreading of grasses (e.g. *Eragrostis aegyptica*, *Phragmites australis*, etc.) could be observed, and these plants became more and more dominant in the area. Altogether 30 macrophytes could be determined (Täckholm 1974).*

* I am very much indebted to the late Professor Vivi Täckholm, formerly Head of the Botanical Department of Cairo University, for determination of the macrophytes collected together with Dr. N. G. Dastane, FAO Expert, during 1972–1973 in the Lake Nasser and Lake Nubia area.

Fig. 7. First seedlings of *Glinus lotoides* arranged in parallel rows by wave action during decreasing water level (January 1970)

Fig. 8. Bushy vegetation in Khor-el-Birba about 5 km from the main Nile Valley near Aswan, at low water level in 1973 (*Tamarix nilotica*, *Rumex dentata* and *Glinus lotoides*)

The colonization of macrovegetation is supposed to have happened in four different ways as spread by (*i*) water current from the south; (*ii*) wind effect mainly from the north; (*iii*) expansion of typical desert vegetation by different means, (*iv*) by animal and human activities (e.g. attached to the feet of waterfowl, the nets of fishermen, boats brought from the Nile, etc.).

It can be concluded that the fauna and flora of the newly formed littoral of Lake Nasser–Nubia have undergone tremendous changes since the formation of this huge man-made lake.

These changes can be well understood as successive processes both on land and underwater, starting from almost lifeless conditions. Formation of life, or better-to-say the settlement of organisms began in this habitat by the appearance of a very unstable, amphibious but single community of aquatic and terrestrial plants and animals hardly to be considered an ecosystem. In a few years this developed into two, more complex and more stable communities by splitting and forming a lacustrine littoral ecosystem interrelated with another purely terrestrial one along the shore.

These two new ecosystems are more and more resembling the original ones of the previous Nile littoral.

However, this process induced only by totally artificial circumstances has already taken ten years and is going to continue most probably for many years, particularly because of the slow stabilizing of the working level of the reservoir even between wide ranges. Several important changes can still be expected, therefore it seems very fruitful from both the practical and scientific viewpoints to continue the studies on this rather unique ecosystem.

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THE ACTIVITY OF THE LIMNOLOGICAL INSTITUTE ON LAKE BAIKAL

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The Limnological Institute of the Siberian Branch of the USSR Academy of Sciences is the centre of investigations on Lake Baikal. This research institute in Siberia was founded some fifty years ago. However, investigations on the lake had begun prior to the foundation of the Institute (Galaziy 1965).

Flora and fauna lists completed at that time revealed some hundreds of unknown species of animals and plants inhabiting Lake Baikal. The bed profile was studied and a bathymetry map was compiled. Palaeolimnological investigations led to the confirmation of our knowledge about the development of lake basins in East Asia and their malacofauna from the early Jurassic period to recent times.

During World War II, investigations were carried out to solve economic problems as well as some others connected with national defence. Proposals for the intensification of fishery on Lake Baikal were put forward; possibilities for using the ice cover of the lake for transport ways were elaborated.

After the war, in connection with the development of the productive forces in Siberia and with the construction of the Irkutsk hydroelectric station, the Limnological Research Institute contributed to the solution of a number of important problems relating to the hydrochemistry, hydrology, and to the thermal and ice regime of the lake. Here we should mention the discussion in 1958 of a suggestion to enlarge the outflow of the Baikal waters through the River Angara. The aim was to quicken the filling of the Bratsk water reservoir. The rejection of this suggestion was a right decision because the decrease in water level of the lake by 5–7 m would have resulted in outcropping of wast areas of coastal shallow waters and in the disturbance of the equilibrium between water and shore. Other noxious effects of this measure would have been landslides on the shores, blocking of spawning ways in rivers used first of all by omul (*Coregonus autumnalis migratorius*) and grayling (*Thymallus thymallus* L.), and also ports would have become shallow or even dry.

At present the Institute's staff counts 355 members (about 120 scientists and engineers), including two corresponding members of the USSR Academy of Sciences, six doctors and forty candidates of sciences. It consists of twelve laboratories, four constant expeditions (Bratsk, Yenisey, Zabaikalye, Baikalo-Amur), a scientific library (with more than 60 000 volumes), a museum and productive sections. There is also a scientific research fleet consisting of ships of different displacement.

Extensive investigations have been carried out at the Institute since its foundation: data have been published on the hydrometeorological regime, water and heat balance of the Baikal and the Bratsk water reservoir, on the hydrochemistry of the Baikal and its tributaries, on the biological productivity of the lake and on human impacts. Geological and limnological studies on the largest Siberian system of Putoran lakes and investigations on the biological productivity of the Ivano-Arakhley lakes in Zabaikalye have been completed. Research on bioproductivity and water quality in reservoirs of the rivers Angara and Yenisey are still in progress. In connection with the Baikal-Amur railway (BAM) construction, the Institute has started complex studies on lake landscapes along the BAM route.

Investigations on the morphology of Lake Baikal performed by echosounding permitted us to describe geomorphological features of the underwater part of the Baikal Basin. Regularities of the rate of slope development in different landscape groups, further sediment transfer along the shore and peculiarities of the orogenesis of the bed profile of the Lake Baikal littoral zone during lithodynamic processes have been established. The relief formation in the River Shartla basin (the Baikal tributary) has been studied by modelling. Preparing of the model of the land-basin material transfer is in progress and a map of exogeodynamic processes in the Baikal depression is being compiled. It is applied for planning the water-safe zone on the Baikal.

New data have been obtained recently on the lithology of bottom sediments of Lake Baikal, presenting unknown facts from the history of the formation of the Baikal depression.

Climatic zonation was described for the Baikal shores and Prebaikalye aimed at their rational economic and recreational exploitation and at planning of water-safe activity. Human impacts on the environment and climate of the Baikal has been studied recently.

As a result of geobotanic investigations, a method has been worked out for the determination of past periods of high water levels of the Baikal as well as of times of formed mud flows, landslide movements, stone avalanches, snow slides, etc. Regularities and limits of water level fluctuations in the lake from post-glacial periods up to the present have been described by this method. Geobotanic maps of the northern Baikal are being compiled.

Regularities for basic hydrometeorological elements in the Baikal water have been studied.

An inclusive study on the ice regime of the lake has been made to describe regularities in the formation and melting of the ice cover. Extensive work has been devoted to characterize the dynamics of the Baikal waters and to the mathematical modelling of admixture distribution to evaluate the influence of sewage of polluted river waters on the biota of the lake.

The hydrochemistry of the lake and of all rivers of the Baikal basin has been studied extensively. The chemical balance of the lake is being checked, and the annual cycles of a series of chemical substances (silicon, iron, phosphorus, organic material, etc.) are monitored; the primary productivity of Lake Baikal and the dynamics of its biogenic elements are studied; and the chemical composition of subsurface and ground waters of the lake coast is determined.

Biological investigations on the dynamics of phyto- and zooplankton resources and on the biology and production of fish fauna permitted us to evaluate, in terms of energetics, the activity of the pelagic ecosystem of Lake Baikal in the creation of organic substance and its transformation into foodstuff. In particular, the ability for producing the main groups of organisms has been determined and quantitative relationships between organisms of different trophic levels have been calculated. It has been established that primarily the littoral zone of the lake, limited by the isobath of 250–300 m, is inhabited by profitable species. From the point of view of the mean annual yield of fish (20–22 kg per hectare), the Baikal shelf zone is similar to that of some other lakes and seas, although this zone is greatly affected by unfavourable anthropogenic influences.

Special attention has been paid to the study of Baikal ciscoes (*Coregonus autumnalis migratorius* Georgi, *Coregonus lavaretus baicalensis* Dybowski), golomyanka (*Comephorus baicalensis* Pall.) and bullheads (*Batrachocottus baicalensis* Berg, *B. nikolskii* Berg, *Cottocomephorus inermis* Jak. and *C. grewingki* Dyb.), playing an important part in the lake's fisheries. Morphology, frequency and weight-size growth, age- and sexual structure, sexual cycles, food, population-size, biomass and production of the main fish species have been studied. Practical suggestions have been worked out for the preservation and increase of resources of most valuable fishes, for their rational economic exploitation and long-term prediction of natural reproduction. Anthropogenic influences caused a decrease in growth, weight, fecundity and ripening time of fish and seal during the last 10–15 years. The role of seal (*Phoca hispida sibirica*) on Lake Baikal, its influence on the fish population and commercial fisheries have been estimated for the first time. Recommendations for the rational exploitation of seal (their number reaches 68 000) have been worked out and introduced into practice. In the Bratsk water reservoir the reproduction of spring-spawning fish has been investigated; the fish fauna during the period of filling up of the reservoir has been analysed and conclusions have been drawn concerning the increase in fish resources.

The maximum depth of Lake Baikal is estimated to be 1620 m, and the absolute (23 000 km^3) and relative volume of the lake fresh water has been assessed (Lake Baikal contains 1/5 of the fresh waters on the globe and above 4/5 of the surface fresh waters in the USSR).

The Baikal Atlas, compiled and published on the basis of these data, describes the main characteristics of this unique fresh water reservoir.

Investigations on the biology of Lake Baikal carried out by the research staff of the Limnological Institute and some other institutions (Zoological and Botanical Institutes of the USSR Academy of Sciences, Moscow and Leningrad Universities, etc.) have improved our knowledge on the flora and fauna of the lake. The number of animals first described in the lake exceeds 300. The lake's invertebrate fauna has proved to be unusually rich. Since 1962 the list of deep-water fauna has been considerably enlarged. At present the list of animals and plants in the Baikal contains more than 2000 species, two-thirds being endemic (up to 1925 only 780 species had been known).

Communities of organisms at the inflow of sewage from the Baikal Cellulose Plant have been controlled since 1967. It has been proved that pollution causes quantitative and structural changes of bottom communities. Based on theoretical values of maximum allowable concentrations of substances and on data about waste materials disposed of with waters into basins, concrete norms for the limitation of harmful substances disposed into the Baikal water have been proposed.

One of the future tasks of the Institute is the automation of collecting and processing of field information by using computers and other special equipment. New laboratories for hydrobiology, biogeochemistry, water ecology, radiobiology, water resources, water dynamics and mapping are to be organized.

The development of productive forces in Siberia and the increase of economic activity in its reservoirs make it imperative to clear up all the influences on fresh water reservoirs, to find measures for neutralizing any negative effects on them and to open new possibilities for increasing their biological resources. The Baikal Limnological Institute plans to study the biology of organisms, the development and functioning of benthos both in Lake Baikal and in other natural reservoirs in Siberia.

Since it is necessary to make observations all the year round on the remote lake systems of Siberia and the Far East, it is necessary to organize research stations on the River Angara (at the town of Bratsk), on the River Yenisey (at the city of Krasnoyarsk), on the rivers Lena, Indigirka, Kolyma, and Taimyr (at Norilsk) and also on Lake Khanka. Research vessels are required for investigations on the rivers Lena, Indigirka and Kolyma, while bathyscaphes or submarines are necessary for the study of the Baikal deep water layers.

An urgent task for the scientists of the Institute is to solve practical questions concerning the utilization and preservation of the natural wealth of the Baikal basin by establishing the qualitative and quantitative relationships in the ecological systems of the lake and its catchment area.

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We recommend

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The participants of the Symposium held at the Biological Research Institute of the Hungarian Academy of Sciences, Tihany, examined the ecosystems of shallow waters. Renowned researchers from several countries took part in the discussion of various topics falling into three main groups: (1) Primary production in shallow waters; (2) Bacterial production and decomposition; (3) Issues of secondary production.